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Recommended Plan for a Comprehensive Solution of the Polynuclear Aromatic Hydrocarbon Contamination Problem in the St. Louis Park Area

Volume II
Appendices A-F

ERT
ENVIRONMENTAL RESEARCH & TECHNOLOGY, INC.
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DEFINITION OF TERMS

Bog	The land immediately south and adjacent to the plant site that received runoff and waste water from the plant site. The bog is bounded by Walker Street to the north, on the east by the route of Louisiana Avenue, on the south by Lake Street and on the west by former railroad embankments. The bog is shown in Figure 2-1.
Benzene Extractable Hydrocarbons	The weight of organic material, per unit volume or mass, obtained on extracting a water or soil sample with benzene. Hydrocarbon in this definition is not meant to be restrictive to material containing carbon and hydrogen, but rather is meant to include all organic material so extracted.
Contaminants	PAH and related compounds that are present in soil or ground water at levels that are clearly above expected background levels for an urban area in the affected media.
Contaminated	Soil or ground water that contains concentrations of PAH and related compounds which are clearly above expected background levels of these compounds for an urban area in the affected media.
Heterocyclic PAH	PAH chemicals with one or more aromatic carbon atoms replaced by nitrogen, oxygen, or sulfur atoms. Unless otherwise stated, alkyl substituted heterocylic PAH are also included by this term.
Hetero-Substituted PAH	PAH chemicals with one or more heteroatomic functional substituted groups attached to the aromatic rings. For the purposes of this term, such functional groups include amines, cyanides, mercaptans, thiols, ketones, ethers, carboxylic acids, alkyl groups, etc. but specifically exclude hydroxyl groups (i.e., phenols).

Modeling Area	An area of 22 miles east-west and 16 miles north-south which is approximately centered on the site area and was modeled for ground-water flow hydraulics as part of this study. The modeling area is defined by the border of the upper map in Figure 2-1.
PAH and Related Compounds	PAH (as defined above) plus related aromatic chemicals that are often associated with PAH in coal tar, soot, petroleum distillates and similar materials. These related aromatic chemicals are by definition limited to heterocyclic PAH, heterosubstituted PAH and phenols (defined elsewhere).
Phenolics	Chemicals measured by standard colorimetric tests for phenols, the current standard test being the 4-aminoantipyrene method. Colorimetric tests typically measure one-ring phenols (phenol and ortho-and meta-substituted phenols and possibly certain para-substituted phenols), and possibly two ring or larger phenols.
Phenols	Chemicals consisting of one or more fused aromatic rings containing carbon and hydrogen with one or more hydroxyl (-OH) groups attached to the ring. Unless otherwise stated, alkyl substituted phenolics are also included by this term.
Plant Site	The land that was formerly the site of the Reilly Tar & Chemical Corporation's creosote wood preserving and coal tar refinery plant. The plant site is bounded on the north by West 32nd street, on the east by Gorham Avenue, Second Street Northwest and Republic Avenue, on the south by Walker Street, and on the West by Pennsylvania Avenue and Oak Hill Park. The plant site is shown in Figure 2-1.

Polynuclear Aromatic Hydrocarbons (PAH)	Chemicals consisting of carbon and hydrogen and containing two or more fused aromatic rings, with each ring consisting of five or six carbon atoms. Unless otherwise stated, alkyl-substituted PAH are also included by this term.
Site	The plant site and bog area together.
Site Area	An area extending approximately 8500 feet east-west and 6000 feet north-south in which the site occupies the northwest corner. The site area is defined by the border of the lower map appearing in Figure 2-1.
St. Louis Park Area	Areas of St. Louis Park and neighboring communities that have been affected by ground water contamination by PAH and related compounds.
St. Louis Park Problem	The overall problems in the St. Louis Park Area related to the contamination of soil and ground water by PAH and related compounds, including in particular the presence of contaminated soil and ground water in the former RT&CC plant site and adjacent bog, the presence of contamination in confined bedrock aquifers underlying the St. Louis Park area, and the present water supply shortage faced by the Cities of St. Louis Park and Hopkins as a result of municipal supply well closures.

DEFINITION OF TRACE CONCENTRATION UNITS

This report makes frequent use of units for expressing trace concentrations. The units used in this report are defined in the table below. In order to avoid misunderstandings over real or suspected typographic errors, an effort has been made to write out all of the units used in this report. Abbreviations are therefore presented below for comparison with other data and reports.

Dimensionless Concentration	Concentration in Water	Concentration in Soil or Other Solid
Parts per million (ppm)	Milligrams per liter (mg/l)	Milligrams per kilogram (mg/kg)
Parts per billion (ppb)	Micrograms per liter ug/l	Micrograms per kilogram (ug/kg)
Parts per trillion (ppt)	Nanograms per liter ng/l	Nanograms per kilogram (ng/kg)

In order to provide a better understanding for the magnitude of these trace concentration units, the following table presents equivalent quantities in more familiar measures of volume, length and time.

Fractional Expression	Volume Equivalent	Length Equivalent	Time Equivalent
1 part per million	1 drop in 26.4 gallons	1 inch in 15.8 miles	1 second in 11.6 days
1 part per billion	1 drop in 26,400 gallons (volume of a typical railroad tank car)	1 inch in 15,800 miles (almost two-thirds of the distance around the earth)	1 second in 31.7 years
1 part per trillon	1 drop in 26.4 million gallons (volume of a 10 acre lake that is 8 feet deep)	1 inch in 15.8 million miles (66 times the distance from the earth to the moon)	1 second in 317 centuries (3 times more distant than the last Ice Age)

APPENDIX A
SITE HISTORY

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A1. INTRODUCTION

This appendix contains historical information about activities at the site of the former Reilly Tar & Chemical Corporation (RT&CC) coal tar refinery and wood treating plant in St. Louis Park, Minnesota. Included in this history is information about the site both before and after RT&CC operations. Most of the descriptions and accounts of the history and operations at the site come from memos and plant records written from 1917 to 1972 when RT&CC owned the property. Information about the site since 1972 is primarily from contractors' reports on their activities on and around the site. This appendix is intended to supplement and provide added details to Chapter 2 of this report.

While much of the information in this appendix does not directly bear on recommendations for a solution to the water and site-use problems in St. Louis Park, it does provide a common background to the problems discussed in this report. Previous reports on the St. Louis Park problem have only briefly addressed the topic of site history and have presented information for which there is only anecdotal evidence. This appendix presents the history of the site based on actual records for which there is ample documentation.

A2. SITE HISTORY PRIOR TO RT&CC PLANT OPERATIONS

As indicated in Chapter 2 of this report, the Minnesota Beet Sugar Manufacturing Company was the first industry to use the site. Details on the operations of the sugar beet refinery are given in two documents located at the Minnesota Historical Society, Main Library in St. Paul, Minnesota. The relevant portions of these documents are summarized below.

The first source of information is a manuscript entitled "St. Louis Park - A Story of a Village" by Norman F. Thomas. An outline of early industrial development in St. Louis Park indicates that the sugar beet company built a refinery in 1897 made up of six large brick buildings on 36 acres. The company employed about 400 men, and in 1905 the corporation was dissolved.

The second relevant document at the Minnesota Historical Society is a book by the U.S. Department of Agriculture entitled "Progress of Beet Sugar Industry, 1902". This book contains statistics on the refinery operation and other details of the economics of sugar beet growing at the turn of the century. Figure A2-1 is reproduced from this source and shows the site in 1902.

A third source of information concerning the sugar beet operations at the site are the pleadings in a lawsuit in 1899 in which the sugar beet company was accused of polluting Minnehaha Creek. A significant statement was made in the pleadings by the sugar beet company. Three million gallons of water per day were used for plant operations. This water usage rate (equivalent to 2083 gallons per minute) is important because it demonstrates that high capacity wells were available and that a significant amount of waste water was probably introduced to the ground water system at this early date.

The wastewater from the sugar beet plant, as described in the pleadings, consisted of beet washing water, water extracted from the beets, refuse syrup, water used to boil the beets (containing beet solids and beet pulp), and occassionaly acids and rinse water used to clean the machinery. No chemical analyses were made to characterize the sugar beet plant effluent. However, based on modern sugar beet operations, it is expected that substantial quantities of phenolics, TOC, COD, and BOD were discharged. Recent investigators have found that wastewater from processing one ton of beets contains about two pounds of total carbon and almost three pounds each of BOD_5 and COD (Brenton and Fischer 1970). As a comparison, the sugar beet refinery in 1902 processed some 32,000 tons of beets (USDA 1902).

A3. SITE HISTORY DURING RT&CC OPERATIONS

This section discusses the time period from 1917 to 1972 when RT&CC owned the site. The section is organized into three parts, the first describes the plant layout, the second describes the various refining and treating operations, and the third describes various hydrocarbon emissions from the plant to the soil and ground water systems.

PILE OF BEETS BY RAILROAD TRACK TOGETHER WITH FORCE EMPLOYED HANDLING BEETS, MINNESOTA SUGAR COMPANY, ST. LOUIS PARK, MINN.

Figure A2-1 Photograph of Site, 1902 (Reproduced from "Progress of Beet-sugar Industry. 1902" U.S. Department of Agriculture)

A3.1 Plant Layout

Figure A3-1 is a reduced scale map of the site based on a 1944 blueprint of the general plot plan. This figure also identifies buildings located on the site. The major buildings to note are the refinery (5-9) the by-products building (13) which was torn down in the 1950's, and the wood treating cylinder building (25). These three buildings housed the major process units of the plant.

Oblique aerial photographs of the site, which were apparently taken in the 1930's and the 1950's (RT&CC files), are reproduced in Figures A3-2 and A3-3, respectively. The site map in Figure A3-1 is helpful in reviewing these aerial photos. Figure A3-2 shows the site in the 1930's. The road in the foreground is Walker Street, and the plant is laid out on 80 acres to the north. The large warehouse in the lower right corner of the photo was originally part of the sugar beet operations. Key features of this photo include the wastewater ditch in south west corner of the plant site (the dark line at the left central portion of the photo); most of the plant site is used for the storage of treated and untreated wood; the northern portion of the site appears to be graded, but as yet unused; the surrounding land is open fields to the north and east, swamp to the south, and wooded to the west.

Figure A3-3 shows the site about 20 years after the photo in Figure A3-2. Perhaps the most notable change that took place in those years is the development of the City of St. Louis Park as it grew up on all sides around the plant site. Also, more of the site is being used for wood storage, especially in the northern portion of the site.

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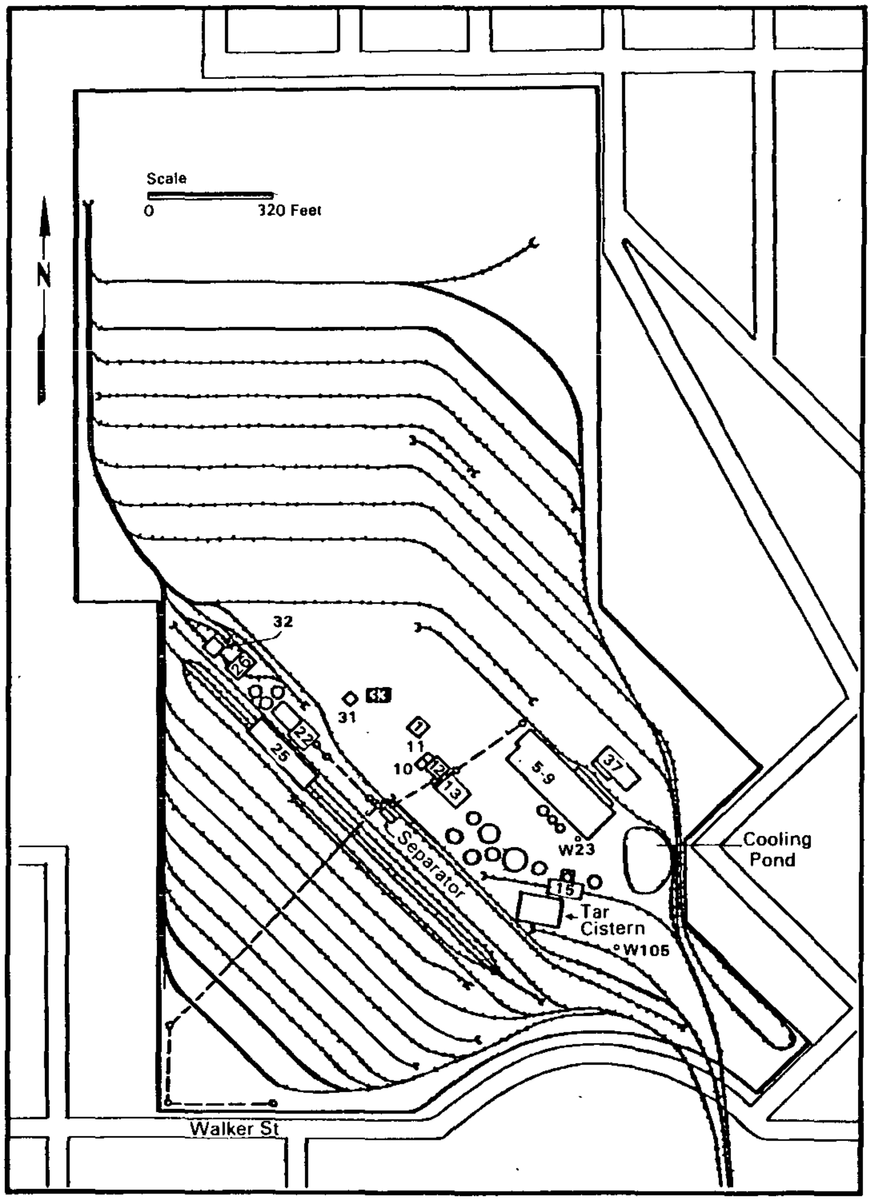

RT&CC PLANT SITE BUILDING IDENTIFICATION

Description	Building No.
Office and Laboratory	1
Refinery	5-9
Blacksmith Shop	10
Locker and Change Room	11
Garage	12
By-Products Building	13
Tar Shed	15
Tank House	21
Boiler House	22
Treating Building	25
Adzing and Boring Mill	26
Garage	.31
Wash Room	33
Tool House	37

EXPLANATION

Narrow Gauge RR Track
Standard Gauge RR Track
Standard/Narrow Gauge RR Track
Sewer Lines

Figure A3-1 Plant Site Drawing From 1944 Blueprint

Figure A3-2 RT&CC Plant, 1930's

Figure A3-3   RT&CC Plant, 1950's

A3.2 Plant Operations

Activities at the RT&CC plant included both coal tar refining and wood treating operations. Refinery operations primarily consisted of the distillation of coal tar in batch stills. The distillate cuts from the coal tar were obtained as the stills were progressively heated to higher temperatures and were called the wet, light oil, middle oil, and heavy oil cuts (see Figures A3-4 and A3-5). Generally the batch tar still operation consisted of charging the still with the crude coal tar, heating the tar to distill off the vapors, and passing these vapors through a condenser to collect the condensed distillate products.

The extent of heating controlled the type of product obtained, such as a road tar, a given type of pitch, or coke. The wet or water cut, consisting of water inherent in the coal tar charge, was disposed by both vaporization and discharging during the early years of plant operation. In later years, the wet cut was condensed and treated before disposal. The light oil cut, including light oils separated from the wet cut, was used in the by-product operation. Middle and heavy oil cuts were blended with acid free light oils from the by-product operation to make creosote preserving oil (see Figure A3-4).

Plant Flowsheets

Figures A3-4, A3-5, and A3-6 are plant flowsheets, which probably represent the coal tar refinery operation in the 1930's and 1960's and the 1920's to 1930's by-product operation, respectively. A discussion of these figures aids in understanding these plant operations.

Figure A3-4 shows the probable flow diagram for the 1930's refinery operation. A noteworthy unit operation unique to the 1930's was the unit #16 isothermal still. The purpose of this still was to perform a mild pyrolysis on water gas tars to give them characteristics similar to those of the coke oven tars predominately used in the refinery. The tar remaining, after being processed in the still, was transferred back to the coke oven tar storage for subsequent processing in the tar distillation batch stills.

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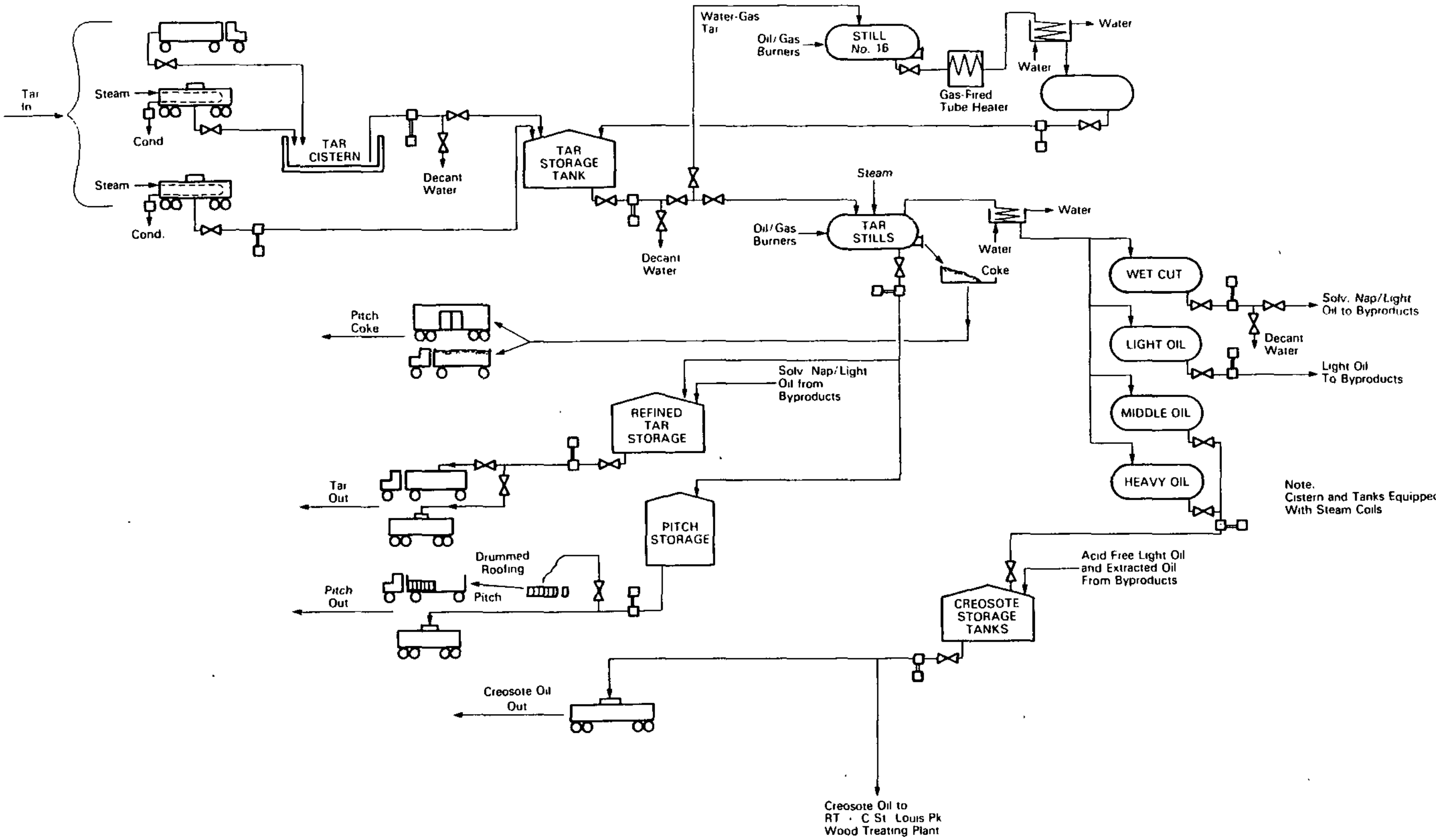

Figure A3-4 Probable St. Louis Park Tar Refinery Operations and Equipment Flowsheet: 1930's

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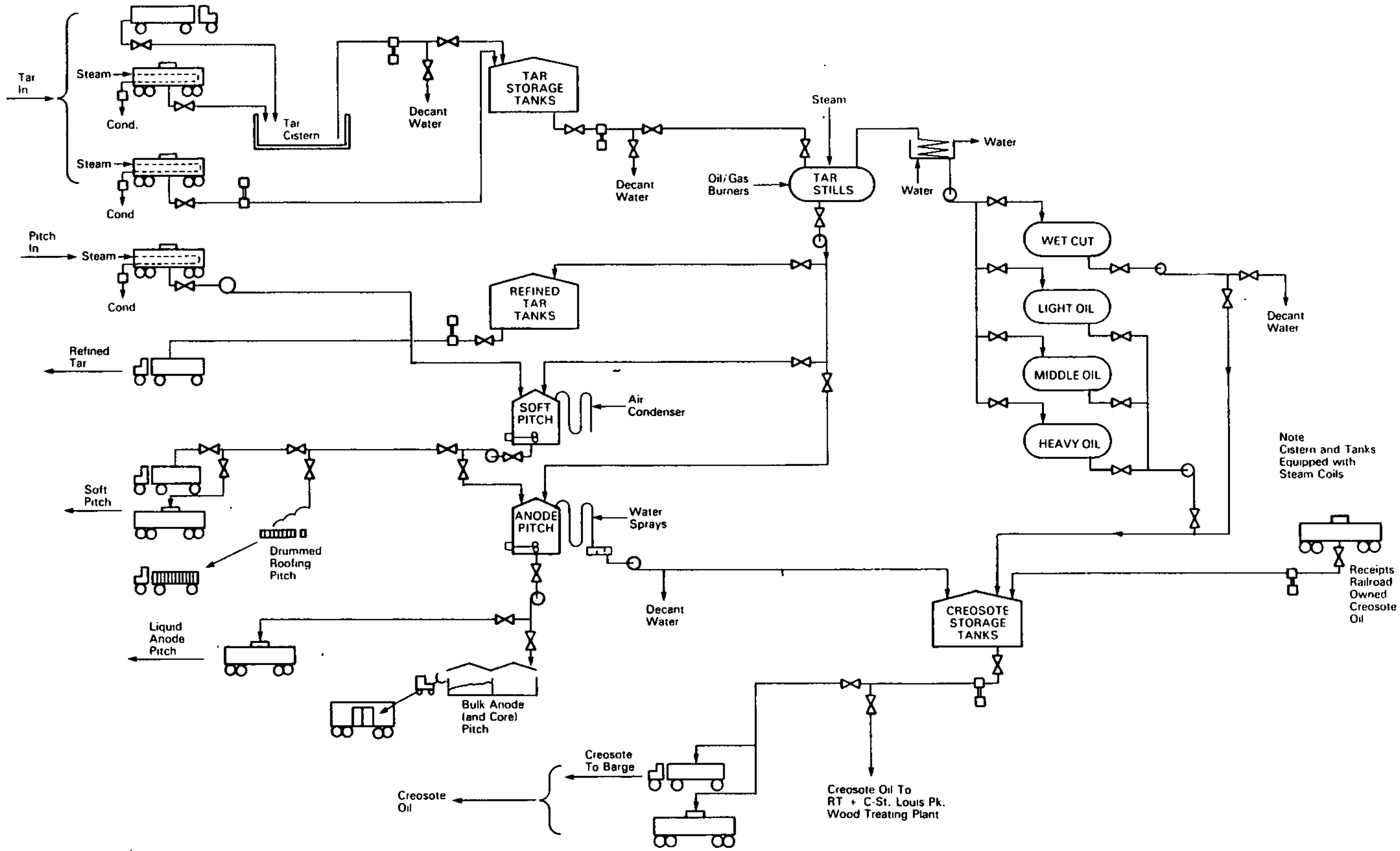

Figure A3-5 Probable St. Louis Park Tar Refinery Operations and Equipment Flowsheet: 1960's

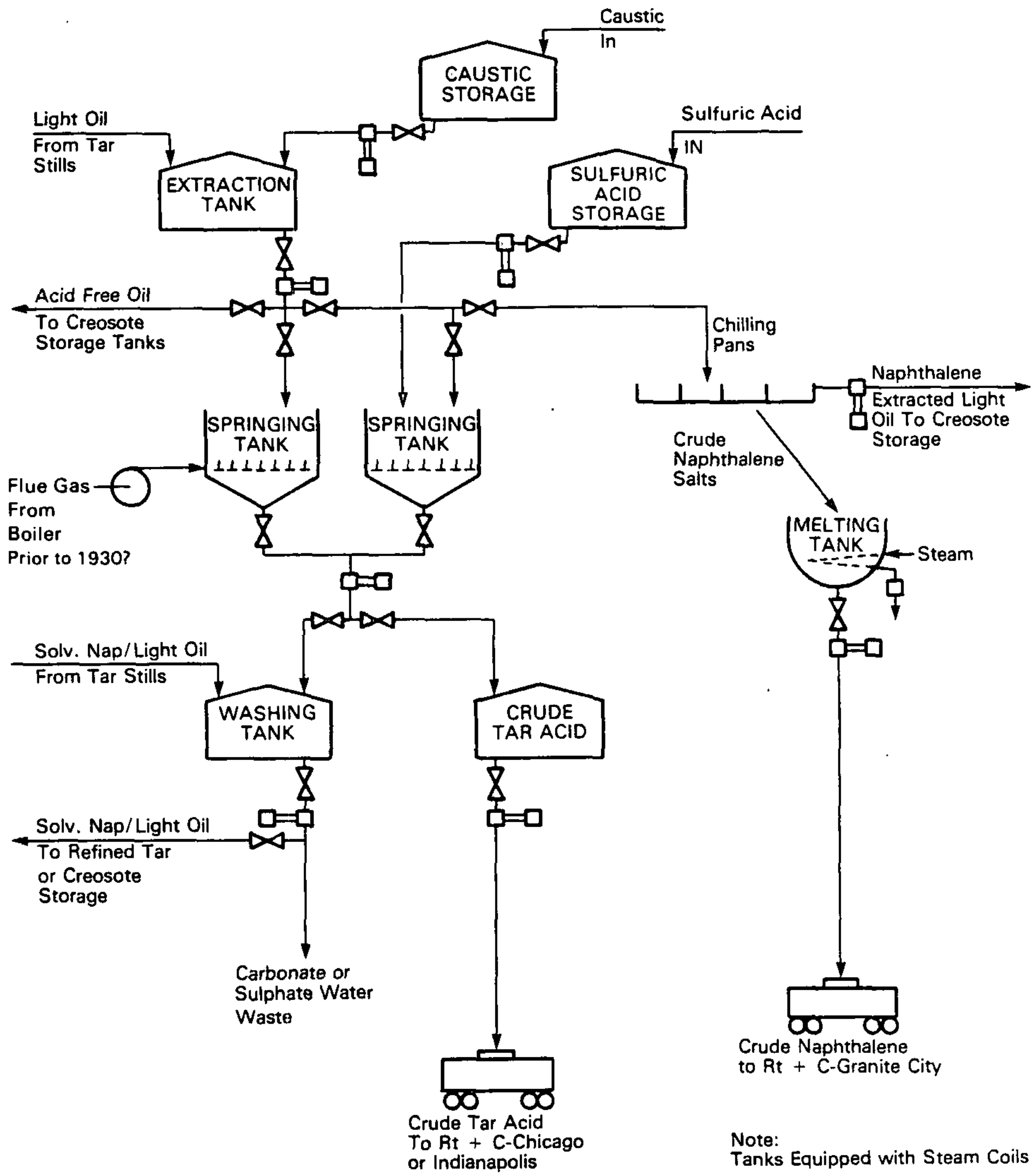

Figure A3-6 Probable St. Louis Park Tar Refinery/By-Products Department Flowsheet: 1920's & 1930's

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The products obtained from the residues of the tar distillation were refined tars, roofing pitch, and coke. The distillate cuts from the still would pass through a vapor condensor to receivers. Light oils and solvent naphtha were used at the by-products plant to produce crude chemical products. The middle and heavy oil cuts were blended with acid free light oil to make the wood preserving creosote oil. This creosote oil was both sold and used at the on-site woodtreating plant.

Figure A3-5 represents the probable refinery operation during the 1960's. The general processing scheme shown is similar to that of the 1930's, although there are significant differencs. The isothermal still was no longer used because the plant only used coke oven tar in the tar stills in the 1960's, eliminating the need for a water gas tar conversion process. Also the total distillate, except for the wet cut, was blended into creosote oil. The major refinery products after the closing of the by-products operation in about 1950 were soft pitch, anode pitch, refined tars, and creosote oil. It is also interesting to note that air and water condensors were placed on the pitch storage tanks in order to alleviate air pollution problems created by escaping pitch vapors.

The probable by-products flow diagram for the 1920's and 1930's is shown in Figure A3-6. Light oil from the tar stills was extracted with caustic to produce an acid free oil and a solution of tar acid sodium salts called sodium carbonate or carbolate solution. It was estimated that 24,000 gallons per year of 70 percent caustic was used in this operation (Holstrom 1943). The carbolate solution from the extraction tank was then processed in springing tanks using sulfuric acid, which liberated the acids and produced sodium sulfate. Edwards estimated that 15,000 gallons of sulfuric acid were used each year at the St. Louis Park plant (Edwards 1930). The acid free oil from the extraction tank was blended with the creosote oil and the crude tar acids were sold. A portion of the extraction tank solution was sent to chilling pans where naphthalene salts would crystallize out of the solution. However, this naphthalene recovery system may never have been used at the St. Louis Park facility (Holstrom 1940).

The sodium sulfate solution obtained from the tar acid springing process was washed with solvent naphtha or light oil to remove any emulsified oils. Starting in 1941, the resulting sodium sulfate wastewater stream was washed with neutral oil to remove phenolic contaminants before discharge to the plant wastewater system. Prior to 1941, the sulfate wastewater was directly discharged.

Crude tar acids were the primary products from the springing operation, and they were shipped to other RT&CC plants that had the capabilities to process these acids into refined products. There is mention made in early plant records, however, that tar acid distillation was performed as part of the by-products operation (Courtney 1938a, Danz 1938a). This distillation was apparently performed simply to remove excess water from the crude tar acids.

Wood Treating Operation

The wood treating operation consisted of two major steps, first the pretreatment of the wood and second the preservation or wood treating step. A variety of wood materials was treated at the RT&CC plant including cross ties, switch ties, piles, poles, lumber, and miscellaneous posts, wood blocks, etc. The pretreatment step included curing and preparing the wood for preservation. The curing operation at the plant consisted of air seasoning, incising, adzing, boring, framing, and final water removal from the wood by treatment with hot creosote (Horner 1942). For air seasoning, the white ties were stacked in the northeast portion of the site and untreated poles and bridge timbers were stacked to the west. After the ties were stacked an anti-splitting device was applied to both ends. Prior to preservative treatment, ties were adzed and bored, and fir timbers were incised to permit penetration of the wood preservative. The seasoned wood was then loaded on trams and pushed into one of three cylinders. The final pretreatment step, used in removing water from the wood, was the hot creosote/vacuum method, sometimes referred to as the Boulton operation (Horner 1942). This method consisted of charging the wood to the treating cylinder, filling the cylinder with hot creosote, (which heated the wood to vaporize its inherent moisture) and then pulling a vacuum to draw the water vapor out of the wood without damaging the wood (Horner 1942).

The actual preserving operation took place in one of three treating cylinders, which were 6 feet in diameter and 176 feet long. Each cylinder held 21 or less trams depending on tie length, or 756 seven inch by eight inch sawed ties per charge. Pressure treating processes were used at the plant. The Rueping pressure treating process was used (Holstrom 1934, Finch 1961). In addition, the Lowry process was probably used on occasion as well.

The Rueping method was the primary wood preserving method used at the plant in 1961 (Finch 1961). The Rueping method consisted of the following: (1) air was introduced to the cylinder and a desired pressure maintained; (2) preservative was added to the cylinder; and (3) the pressure was released and a vacuum drawn which forced some of the preservative out of the wood. This allowed for greater recovery of oil.

The Lowry method consisted of the following: (1) after the vacuum from the hot creosote water removal step was released, the hot preservative solution was pressurized at 150 to 200 pounds per square inch and held until a specified absorption was obtained; (2) the oil was drained; and (3) a vacuum was drawn to remove any excess oil from the wood. After treatment the wood was removed from the cylinders and shipped to customers or stored in the tie yard.

A3.3 Plant Discharges

Two general types of discharges occurred from the plant, 1) wastewater and 2) miscellaneous discharges, such as leaks, spills, and storm water run-off. The following descriptions of these discharges are based on plant and company records.

A3.3.1 Wastewater

Wastewater sources from plant operations were described at various times during the plant's history in plant and company documents. Selected excerpts from key documents describing plant wastewater sources are quoted below to give a sense of the changing sources throughout the plant's history. Not all of the sources identified are process related but are included here as part of the complete quote.

In a memo dated July 27, 1938, W. J. McLellan summarized plant wastewater sources as follows:

"Waste material going to our drainage ditch comes from the following sources:

1. Water from benzol separating tanks (wet cut water)
2. Water from tar acid distillation
3. Free water separating from storage tanks (including tar cistern water)
4. Sulphate water resulting from springing of carbolate
5. Blow-off water from boilers at Creosoting Plant
6. Surface drainage water resulting from rain and melting snow."

As a result of a plant inspection on July 13-20, 1954, J. A. Lauck summarized the plant wastewater sources in a 10/12/54 memo to H. R. Horner as follows:

"Plant waste consists of:

1. Surface water
2. Tank farm trench water - surface water, steam condensate, oil and tar spillage.
3. Water from oil-water cut in refinery.
4. Cooling water from refinery and air compressors
5. Laboratory sink
6. Boiler blowdown
7. Water from tank 5 at treating plant."

Note that this listing no longer includes sources from the by-products operation (sources 2 and 4 in July 1938 listing), which had been discontinued by this time. Sources 4 and 5 above were not listed in the July, 1938 memo, but presumably did exist at that time. Sources 1 and 2 presumably include storage tank and tar cistern water. Source 7 is new due to installation of a separating settling tank for oil recovery from the steam condensate, drippings and water vapor from the treating cylinders.

Various documents concerning wastewater in the late 1960's and early 1970's mention the following wastewater sources:

1. tar cistern water*
2. still distillation water (wet cut)*
3. boiler blowdown*
4. treating cylinder wastes*
5. storm water run-off**

Storm water run-off included site and surrounding area run-off as well as run-off that was pumped on to the site by the city of St. Louis Park. Refinery and air compressor cooling water and non-contact steam condensates from tank heating should presumably also be included on this list.

Wastewater Flow Rates

There is a variety of information available in various company and plant records on the quality and quantity of the wastewater and total effluent resulting from the plant operation.

Table A3-1 lists the wastewater flow rates reported in various documents in the plant and company files. Most of the reported flow rate data are from the 1938 to 1940 period. The reported data are fairly consistent in indicating the relative sizes of the various

*Finch 1971
**Lesher 1968.

TABLE A3-1

REPORTED FLOW RATES FOR PLANT WASTEWATER

Stream	Date	Flow Rate: As Originally Reported	Flow Rate: Converted to gpm[h]	Reference
Wet Cut	5/38	400 gal/working day[j]	0.28	1
	1938	4800 gal/month[a]	0.11	2 & 3
	1940	2400 gal/week[b]	0.48	4
	1970	300 gal/day	0.21	9
Sodium Sulfate Liquor	5/38	1500-3000 gal/week	0.15-0.30	1
	1938	8000 gal/month[k]	0.18	2 & 3
	1940	3500 gal/week	0.35	4
Tar Acid Distillation Water	5/38	1000 gal/month	0.023	1
	1939	900-1000 gal/month[c]	--	3
	1940	1200 gal/month	0.028	4
Drainage Ditch	5/16/38	200 gpm[i]	200	1
	5/19/38	150 gpm[i]	150	1
	11/7 & 9/38[d]	∿ 50 gpm	∿ 50	5
	11/7 & 9/38[d]	47 gpm	47	6
	U[e]	∿ 40 gpm	∿ 40	7
	1/25-26/41[e]	∿ 35 gpm	∿ 35	7
Water to City Sewer	1969	154.703 ft^3/month	27	11
Water to Separator[f]	1970	200 gpm maximum	--	10
		< 100 gpm normally	--	
Plant Discharge[g]	1971	4800 gal/day	3.3	8

REFERENCES FOR TABLE A3-1

Reference Number	Reference
1	Minnesota Department of Health (MDH) Draft Report by Kempe, May 1938.
2	Memo from McLellan to Edwards, 7/27/38.
3	Memo from Danz to Courtney, 9/28/38.
4	Memo from Mitchell to Larkin, 5/1/40.
5	Memo from Danz to Courtney, 11/10/38.
6	Memo from Courtney to Edwards, 11/18/38.
7	Graphs of "Republic Creosoting Co. St. Louis Park Plant Flow of Drainage Ditch" attached to a copy of the 1938 MDH report showing hourly flow rates for January 25-26, 1941 (1) Unit of flow cannot be read on graph but is assumed to be gpm.
8	Memo from Boyle to Finch, 2/5/71, reporting information required for U.S. Army Corps of Engineers Waste Materials Permit under Federal Refuse Act.
9	April, 1970 Minnesota Pollution Control Agency (MPCA) report.
10	Hennessey to Finch, 9/30/70.
11	Anonymous handwritten calculation on St. Louis Park plant paper, 12/8/69, which calculates "sewage charges" based on average cubic feet of unidentified water from 12/20/68 to 10/23/69.

NOTES FOR TABLE A3-1

(a) Based on "average production being 100 gallons per still, four stills being operated three time per week which gives 1200 gallons per week or 4800 gallons per month." (ref. 2).

(b) Described as "water with light oils" (ref. 4).

(c) "Normally we will produce sufficient crude acid to make 2-4000 gallon still runs per month, which will yield from 900 to 1000 gallons of water, indicating the crude acid contains approximately 12% water, which is a normal condition" (ref. 3).

(d) Average flow "over the period of 9 a.m. to 3 p.m. on 11/7/38 and 11/9/38" (ref. 6).

(e) Approximate average of flow rates measured hourly (presumably measured with weir installed in 1938) over 24-hour periods. 35 gallons per minute flow was very steady throughout the 24 hour period, while 40 gallons per minute flow was generally steady at 30-35 gallons per minute except for surges of 80 gallons per minute at 9 p.m. and midnight with 50 gallons per minute flow in between these two spikes.

(f) Basis for contemplated design of new separator.

(g) Basis for this low flow rate is unclear.

(h) Based on continuous flow throughout the originally reported time period.

(i) Reference 1 reported that "most of this flow was ground water and surface runoff."

(j) Reference 2 implies that this figure is too high.

(k) The discharge frequency for this batch wastewater discharge was probably 2 or 3 time per month, based on statement in 11/29/38 Danz to Courtney memo that one discharge amounted to 3500 gallons.

waste streams. The tar acid distillation water and sodium sulfate liquor were batch discharge wastes (a few discharges per month - see footnotes c and k to Table A3-1) of about 1,000 and 6000-12000 gallons per month, respectively (about 0.02 and 0.2 to 0.3 gallons per minute on a continuous basis, respectively). The wet cut wastewater was also a batch discharge, although on a more frequent basis (about twelve times per week in 1938 per reference 2 - see footnote a to Table A3-1), amounting to about 4,800 gallons per month (or 0.1 gallons per minute on a continuous basis).* The 1970 estimate for the wet cut is somewhat higher at 300 gallons per day or 0.2 gallons per minute on a continuous basis.

Plant Wastewater Flow Diagrams

Based on the evolution of wastewater treatment methods at the plant and the description of the wastewater sources and flow rates, general flow diagrams of the plant wastewater generation, treatment, and disposal system have been prepared for various periods in the plant's history. Figures A3-7, A3-8, and A3-9 show such diagrams for three distinct periods in the plant's history: 1917-1939, 1940-1954, and 1955-1972. While the division points between these periods are not precise, the general result of there being three different eras in the plant operation from a wastewater perspective is felt to be legitimate and important based on the following distinctions:

- From 1917 to about 1939 no wastewater treatment was employed;

- From about 1940 to 1950 wastewater treatment was employed and the by-products operation was in effect; and

*The 5/38 MDH estimate of 6,000 gallons per week is high in that the plant's average discharge was 100 gallons per still and four stills were operated three times per week. The 1940 estimate has been discounted because it appears as if the light oil cut may be included in this estimate.

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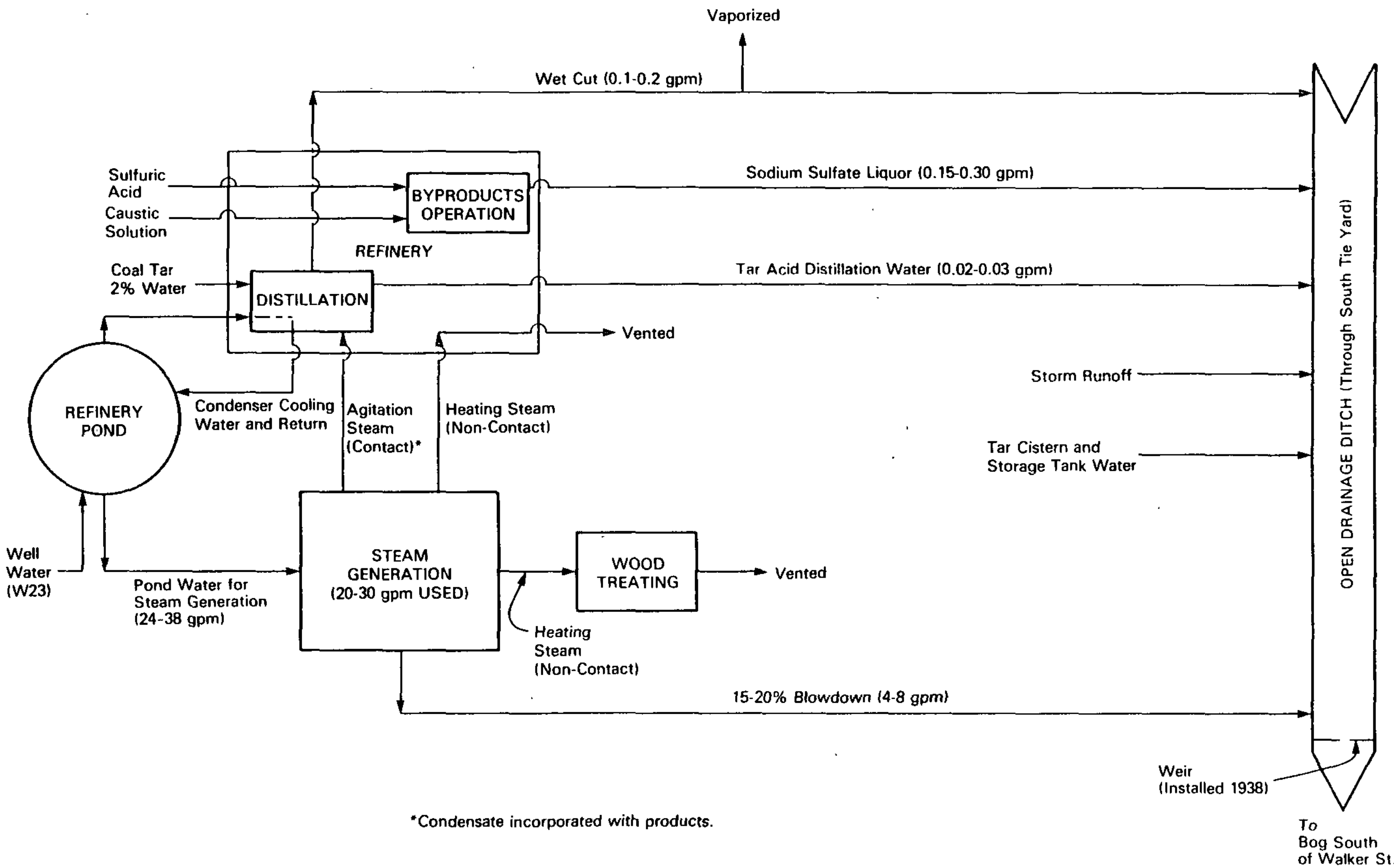

Figure A3-7 Reconstructed St. Louis Park Plant Wastewater Flow Diagram: 1917-1939

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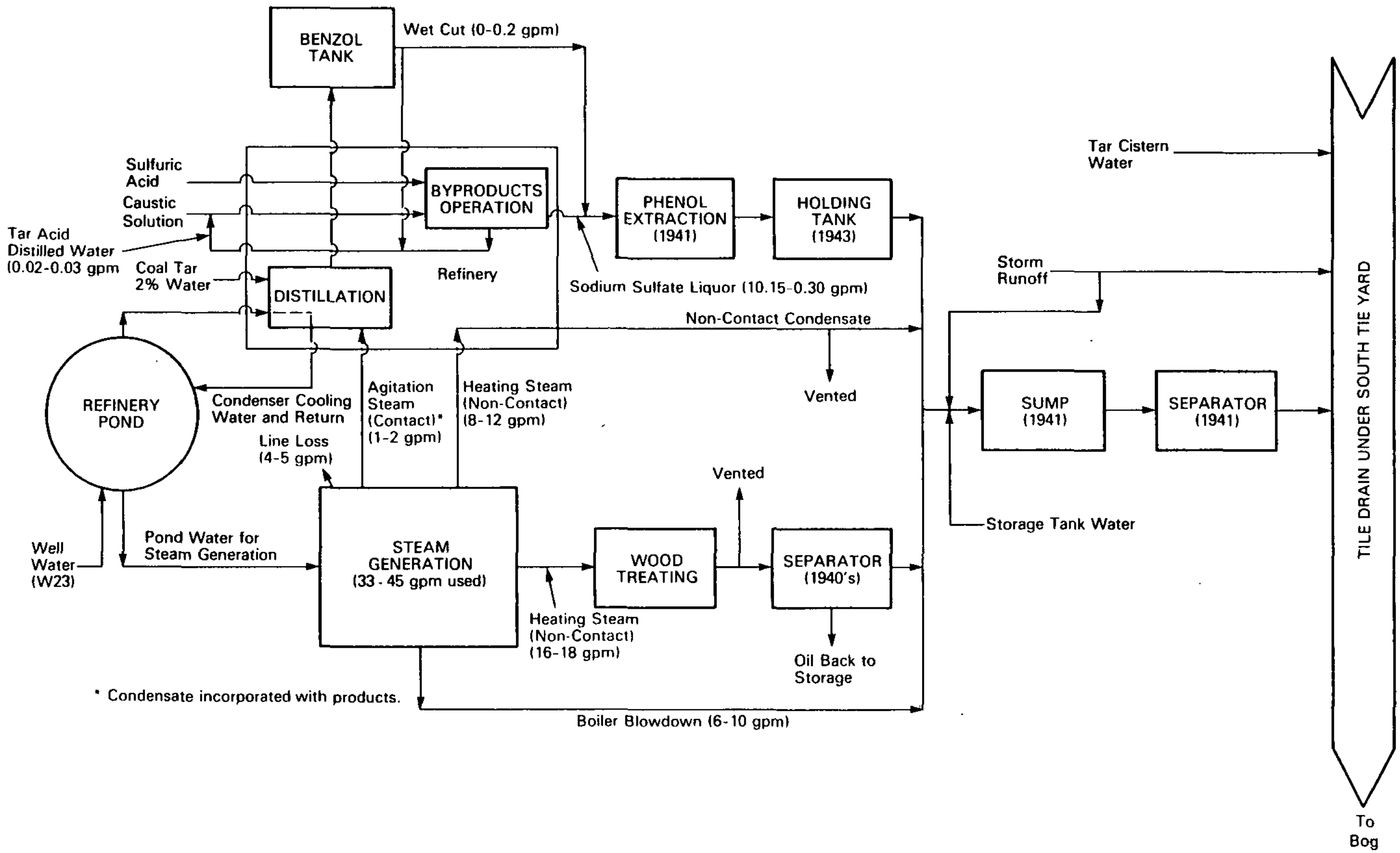

Figure A3-8 Reconstructed St. Louis Park Plant Wastewater Flow Diagram: 1940-ca. 1950

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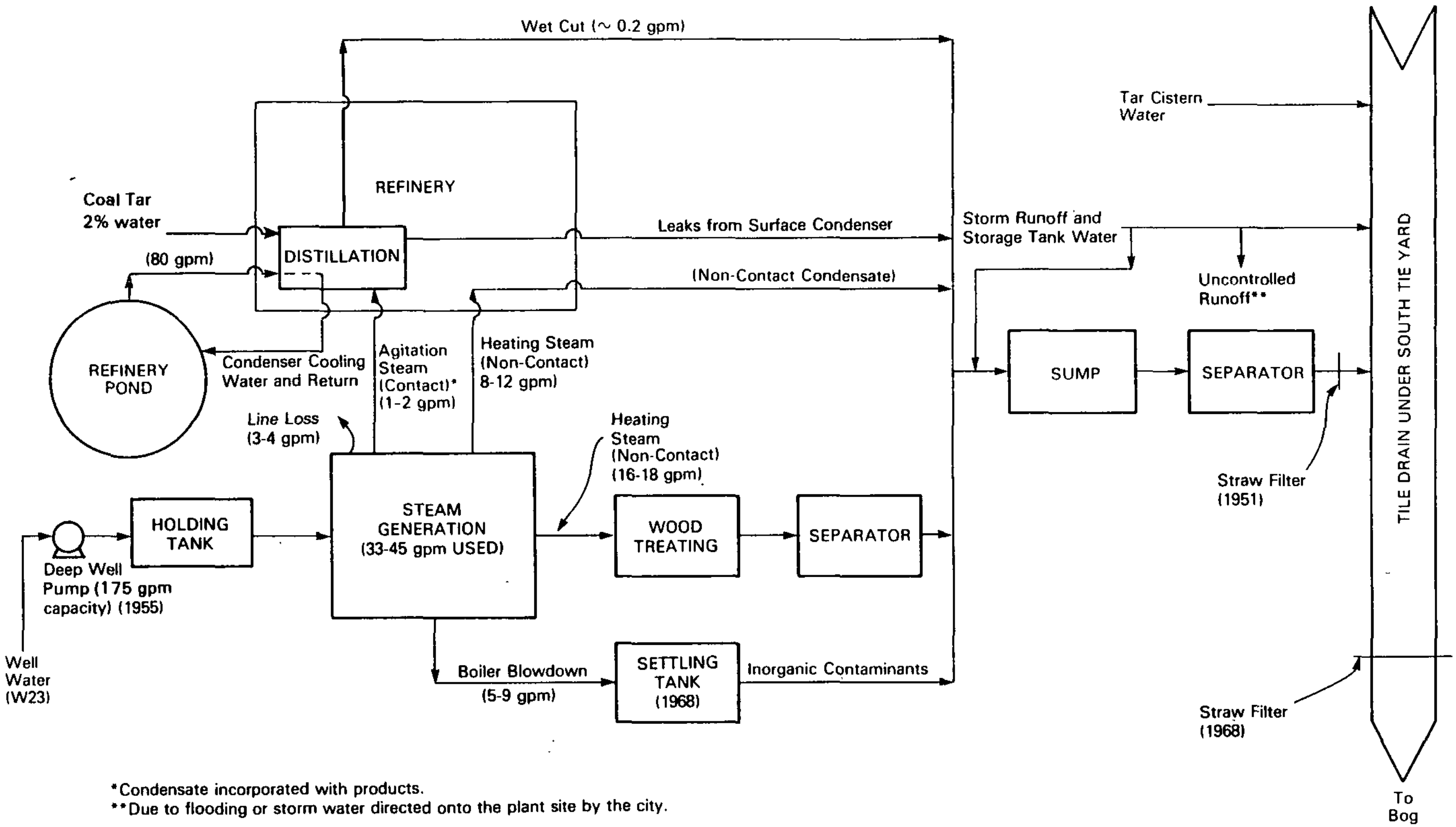


*Condensate incorporated with products.
**Due to flooding or storm water directed onto the plant site by the city.

Figure A3-9 Reconstructed St. Louis Park Plant Wastewater Flow Diagram: ca. 1951-1972

- From about 1950 to 1972 wastewater treatment was employed but without by-products waste streams.

Further discussion of the plant wastewater operations in these periods is provided below.

1917-1939

As shown in Figure A3-7, from the opening of the plant in 1917 to about 1939, all plant wastewaters from the refinery, by-products, and wood treating operations were discharged untreated to an open ditch which carried them through the south tie yard to the plant discharge point at Walker Street. From here the wastewater flowed into the series of bogs and swamps south of the plant. Up until about 1930 the water could flow through this area into Minnehaha Creek, but this route was restricted when a culvert under Lake Steet collapsed (Minnesota Department of Health 1938a). Much of the steam used in the plant during this period was directly vented to the atmosphere after use.

Little information on the quantity and quality of the various wastewaters is available for this period except in 1938. What little flow rate information is available is summarized in Figure A3-7, although it must be cautioned that these data may not be representative of operations in earlier years.

1940-1950

Major wastewater treatment units were installed in 1940 and 1941. A phenol extraction tank and separator were installed to treat various wastewaters, as shown in Figure A3-8. An oil/water separating tank (tank no. 5) was also installed at the wood treating plant during this period. Tar acid distillation water was no longer discharged but was reused in the plant. Available information on wastewater flowrates for this period is summarized in Figure A3-8.

1951-1972

The wastewater treatment units installed in 1940 and 1941 continued in use during the remainder of the plant life except for the phenol extraction unit, which presumably ceased operation when the by-products operation was closed in about 1950. In addition, various changes to the wastewater treatment system and plant operations were made during this period to improve wastewater quality. Straw filters were added to the separator outlet in 1951 and in the Walker Street drainage ditch in 1968. Water from W23 was directed to a holding tank instead of the cooling pond in 1955, and a separate boiler blowdown settling tank was added in 1968. Information on wastewater flow rates during this period, is summarized in Figure A3-9.

Wastewater Quality

Data on the quality of various plant wastewaters were obtained by direct measurement during two different time periods in the plant's operating history: 1938 to 1941 and 1968 to 1972. Wastewater measurements during these periods resulted from an inspection by the Minnesota Department of Health (1938b) and general environmental impact concerns by the State of Minnesota and the City of St. Louis Park in the late 1960's. Measurements were made by a variety of organizations, including RT&CC, the Minnesota Department of Health, the Minnesota Pollution Control Agency, and Twin Cities Testing Laboratory (hired by RT&CC).

Table A3-2 summarizes all of the available analytical data for oil and grease and phenolics of the plant discharge. These samples were generally collected in the drainage ditch at the plant property at Walker Street. The results show that phenolics ranged from 0.5 to 1,600 milligrams per liter and oil and grease ranged from 1 to 2,300 milligrams per liter. These wide ranges demonstrate the variability in discharge quality, and they also reflect differences in sampling and analytical methods.

TABLE A3-2

QUALITY OF DRAINAGE DITCH WATER LEAVING
RT&CC PLANT SITE

Date Sampled or Reported	Laboratory/Agency (a)	Concentration Milligrams per liter: Phenolics	Concentration Milligrams per liter: Oil & Grease
5/19/38	MDH	50	-(b)
7/38	UNKNOWN(c)	62.5	-
8/27/38	RT&CC(c,d)	109	158
11/7/38	RT&CC(d)	-	106
11/9/38	RT&CC(d)	-	436
11/7 - 9/38 Composite	RT&CC(c,d)	112	130
11/18/38	RT&CC(c,d,e)		
11:20 am		213	784
1:30 pm		1180	2260
2:30 pm		1640	1580
3:30 pm		1050	1360
5/10/40	SLP(c)	127	-
	RT&CC(c)	146	-
6/6/40	SLP(c)	30	-
	RTC(c)	82	-
5/9/68	TCT	20	207
6/13/68	MPCA	130	-
6/24/68	TCT	13	25
8/1/68	MPCA	140	-
11/15/68	TCT	1.4	3
1/20/69	TCT	0.5	1
12/21/69	TCT	0.56	1.6
4/14/70	MPCA	150	-
4/18/70	UNKNOWN	1110	-
9/3/70	TCT	3.5	-
10/29/70	CWR	9.8	-
10/29/70	MI	13.0	-
12/9/71	MPCA	140	140
7/21/72	MSB	7.5/11.5	41/41

a) The following laboratories or government agencies conducted or reported results:

MDH - Minnesota Department of Health
RT&CC - Reilly Tar & Chemical Corporation
SLP - St. Louis Park
TCT - Twin Cities Testing
MPCA - Minnesota Pollution Control Agency
CWR - C. W. Rice Company
MI - Mellon Institute
MSB - Metropolitan Sewer Board

b) A dash means parameter not analyzed.

c) Phenolics determined by Chapin method.

d) Oil & Grease determined by extraction with ether.

e) Samples collected while hot sulfate water was being discharged from plant.

The time-series data of November 18, 1938 illustrate one aspect of varying effluent quality by showing the effect of discharging hot sodium sulfate liquor, a batch process wastewater. The large increase in waste loadings shown by the time-series data is not unexpected, given the high phenolics concentrations in this stream and the likelihood of settled oils and tars on the bottom of the ditch being solubilized by the surge of hot effluent in the ditch.* Dilution from storm water run-off flows and the batch or semi-batch nature of many process wastewaters (e.g., wet cut water and tar acids distillation water) also affects wastewater quality variability.

The importance of sampling and analysis variations in interpreting the data of Table A3-2 is evidenced by comparing the phenolics results by the RT&CC St. Louis Park and Indianapolis Laboratories for the same samples (May 10, 1940 sample differed by 15 percent and June 6, 1940 by a factor of 2.7) and by comparing the phenol results of Minnesota Pollution Control Agency (MPCA) and Twin Cities Testing (TCT) for a series of ditch effluent samples collected at different times in 1968-1971. The average MPCA phenolics result was 140 $\pm$ 8 milligrams per liter for four samples (excluding a high value of 1,110 milligrams per liter) compared to 6.5 $\pm$ 8.1 milligrams per liter of phenolics for six samples by TCT. Further details on sample collection times and analysis procedures might explain these discrepancies.

Given the limitations of the data, Table A3-3 does indicate a general improvement in the quality of the plant discharge water from the 1938-40 period to the 1968-72 period. This is in agreement with the fact that significant pollution control steps were added in 1940 and 1941. These steps included installation of a phenol extraction unit in 1940, which reduced phenolics levels in the wet cut wastewater, and installation of a settling basin for the plant

*Analysis of the ditch mud in August 1938 showed a high oil and grease content (about 5 weight percent) for the mud.

TABLE A3-3

AVERAGE OIL AND GREASE AND PHENOLICS RESULTS FOR PLANT DISCHARGE WASTEWATER(a)

Averaging Basis	Laboratories	Milligrams per Liter Mean ± Std. Dev. for (n) analyses Phenolics	Oil & Grease
1939-1940 analyses(b)	MDH, RTC, SLP	90 ± 40 (8)	208 ± 154 (4)
1968-1972 analyses	MPCA(c)	140 ± 8 (4)	140 (1)
	TCT	6.5 ± 8.1 (6)	48 ± 90 (5)
	C.W. Rice, MSB, Mellon Institue,	10.4 ± 2.4 (4)	41 (1)

Explanation

MDH - Minnesota Department of Health
RTC - RT&CC laboratory
SLP - St. Louis Park plant laboratory
MPCA - Minnesota Pollution Control Agency
TCT - Twin City Testing laboratory
MSB - Metropolitan Sewer Board

(a) Averages of results in Table A3-2.
(b) Excluding 11/18/38 time series data.
(c) Excluding 4/18/70 analysis.

effluent in 1941. The settling basin would be expected to reduce oil and grease levels and possibly have some reducing effect on phenol levels as well, given phase partitioning of phenolics between oil and aqueous phases.

A3.3.2 Miscellaneous Discharges

Infrequent and accidental discharges at the plant included flooding at the site due to storm water runoff, (or significant snow melting) spills and leaks, fires and explosions, tank cleanings and sludge disposals, and creosoted wood seepage. Of these discharges, tank cleanings and sludge disposals are believed to have constituted the greatest amount of waste material. The viscous sludges were often used in the northern tie yard to extend and improve roadways. Therefore, some fraction of the oil and grease and phenolics measured in the drainage ditch (see Table A3-2) may be related to other than actual process wastewater.

Major floods at the site have been documented on several occasions including June 11 and August 13, 1956 (Mootz 1956, Lauck 1956), and October 9, 1970 (Justin 1970). Floods probably occurred at the plant that were not documented. Floods became common as the City of St. Louis Park grew around the plant, and much of the area became paved, directing storm water runoff onto the site. Floods were a major probelm at the plant because they left several inches of standing water in the refinery building and in the yard, thus stopping operations. Also, the flood water caused the wastewater system to overflow, rendering it ineffective. It is not known to what extent flood waters carried contaminants from the site.

A4. SITE HISTORY AFTER THE RT&CC PLANT WAS CLOSED

In 1972, RT&CC ceased operations at the St. Louis Park facility and the plant was razed. The razing operation left the site free of buildings, equipment, or other large demolition debris. All below grade tanks, piping and machinery were removed and all basements and low areas were filled in. In general, the site was leveled.

During the period from 1972 to present, the site has been the focus of numerous environmental studies and at least three major construction projects aimed at converting the site to non-industrial uses. A storm sewer was constructed starting in 1973 to control storm water runoff in the area. Part of this project included the installation of a lined holding pond in the southern portion of the site. In 1976, plans were developed to construct condominiums on the site for the general public. The project is called Oak Park Village, and to date, the northern portion of the property has been used for this purpose. Finally, in 1977 plans for the extension of Louisiana Avenue through the site were made. This extension has been completed, although an intersection with Highway 7 has not yet been built due to complications involving the bog to the south of the site. A summary of these and other activities is provided in Table A4-1. In order to present a view of how the site has been modified by these construction projects, a series of maps is provided in Figures A4-1 through A4-6.

A rough estimate of the relation of the former plant layout to the present site can be made by comparing Figure A3-1 with Figures A4-1 through A4-6. In general, the residential structures are located where the northern tie yard of the plant was. While this area was used primarily for wood storage, some disposal or filling took place in this area, and indeed some contaminated soils were excavated during construction of Oak Park Village. Louisiana Avenue cuts through the eastern portion of the site and probably was built directly over the plant cooling pond. The lined storm water pond on the site is located where the southern tie yard used to be, with its northern end extending to or just south of the plant's sump/separator location.

Figure A4-7 is an aerial photograph of the majority of the site taken on November 26, 1982. There was a dusting of snow the day the area was photographed. A topographic map made from these photographs is shown in Figure A4-8. All of the major surface features of the site can be seen on the topographic map including the condominiums, the pond, and the asphalt walkways. A view from atop the stockpiled soil in the southwest corner of the plant site towards the condominiums to the north is shown in Figure A4-9. In addition, Figure A4-10 shows the plant site from atop the stockpiled soil looking south. Most of the site is grass covered with many trees, creating a park-like setting.

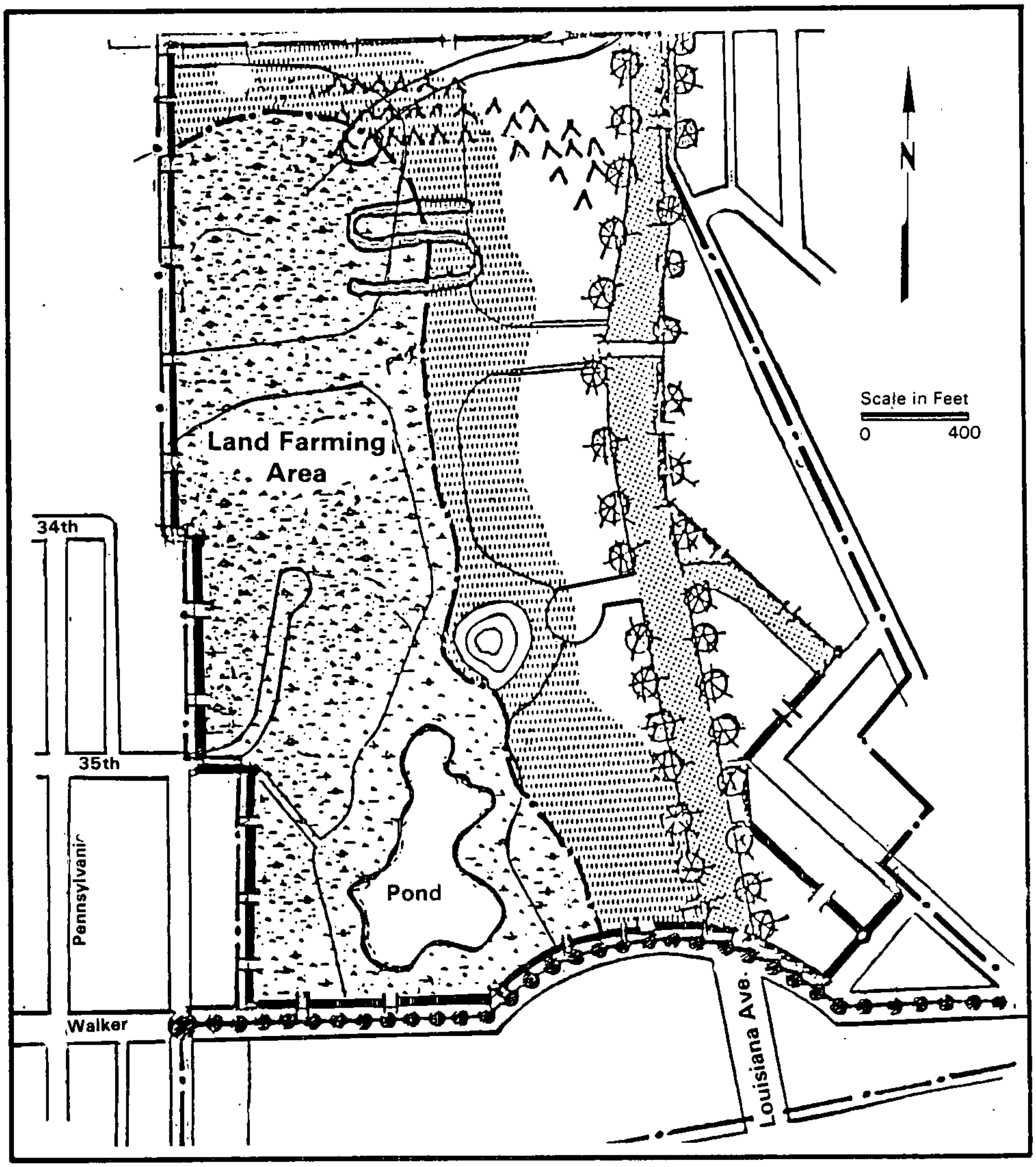
TABLE A4-1

TIMETABLE OF SELECTED ACTIVITIES CONCERNING THE FORMER RT&CC ST. LOUIS PARK PLANT SITE, 1971-1980

Year	Description
1971	Inventories were being emptied 6 soil borings for the City of St. Louis Park Refinery closed Tar Cistern ditch filled
1972	City and RT&CC negotiate on plant clean-up Plant discontinues operation City purchased plant site RT&CC contracted site razing to Carl Bolander & Sons
1973	Microbial soil analysis performed by U. of Minn. Plant growth study by U. of Minn. SLP consultant recommends landfarming 13 soil borings for storm sewer investigation
1974	17 soil borings to evaluate soil and groundwater conditions for storm sewer Hydrogeolic study of site area by Sunde
1975	Serco laboratories study for monitoring program for an NPDES permit City proposed a disposal system for treatment and collection of the contaminated soils and their associated surface runoff water City is granted NPDES permit (April) 2 soil borings for the stormwater retention pond
1976	Proposal to excavate the north end of the plant site to a depth of 8' and move this material to the south end of the plant site for landfarming Development plans for the northern portion of Oak Park Village Barr Phase I report 18 soil borings to identify the extent of visual contamination within the top 10 ft. on the northern portion of the site 4 soil borings to measure amounts of contamination at irregular depths Environmental assessment of Oak Park Village Environmental assessment of Oak Park Village Supplemental Information

TABLE A4-1 CONT'D

Year	Description
1977	MPCA proposes to establish the design and construction of a barrier well system 3 soil borings for Barr Engineering Barr Phase II report Louisiana extention proposal plan for construction
1978	Estimated that 44,000 cubic yds. 1.2 x 10^6 $ft.^3$) of soil was excavated from northern portion of site Status report on Proposed Development of Oak Park Village 6 soil borings for Louisiana Extension soil analysis 4 soil borings to investigate soil conditions for construction of Oak Park Plaza and bowling alley 2 soil borings for USGS to install piezometers for monitoring water levels
1979	MDH put out RFP for remedial measures Townhouse construction
1980	USGS locates W105 City proposes temporary Louisiana Avenue Extension Bowling alley construction, including moving contaminated soil
1981	USGS Report E.A. Hickok & Assoc. Report
1982	Exploratory/remedial work begins on W23, RT&CC plant supply well CH2M Hill begins drinking water treatment study



EXPLANATION

- NDP Acquisition Boundary
- NDP Project Boundary
- Louisiana Avenue
- Walker Watermain
- Grading
- Land Farming
- Trees
  - 9 American Linden
  - 9 Red Maple
  - 14 Marshall Green Ash
- Temporary Leaf Compost Row
- Temporary Storage of Fill

*Reproduced from "Supplemental Information Environmental Assessment Oak Park Village", 6/23/76, Doc. No. 50002992-50003063.

Figure A4-1 Location of Proposed Landfarming Area

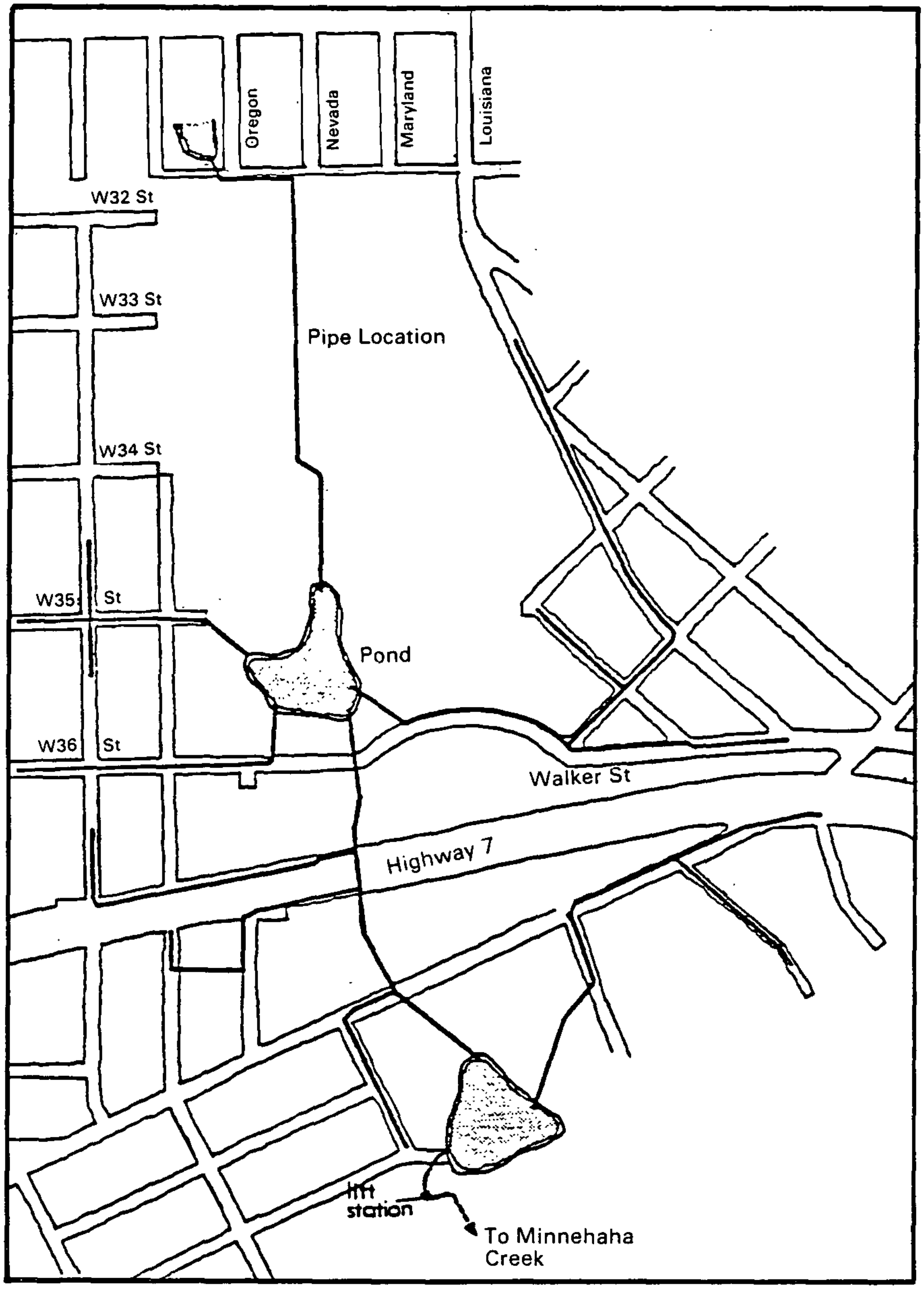
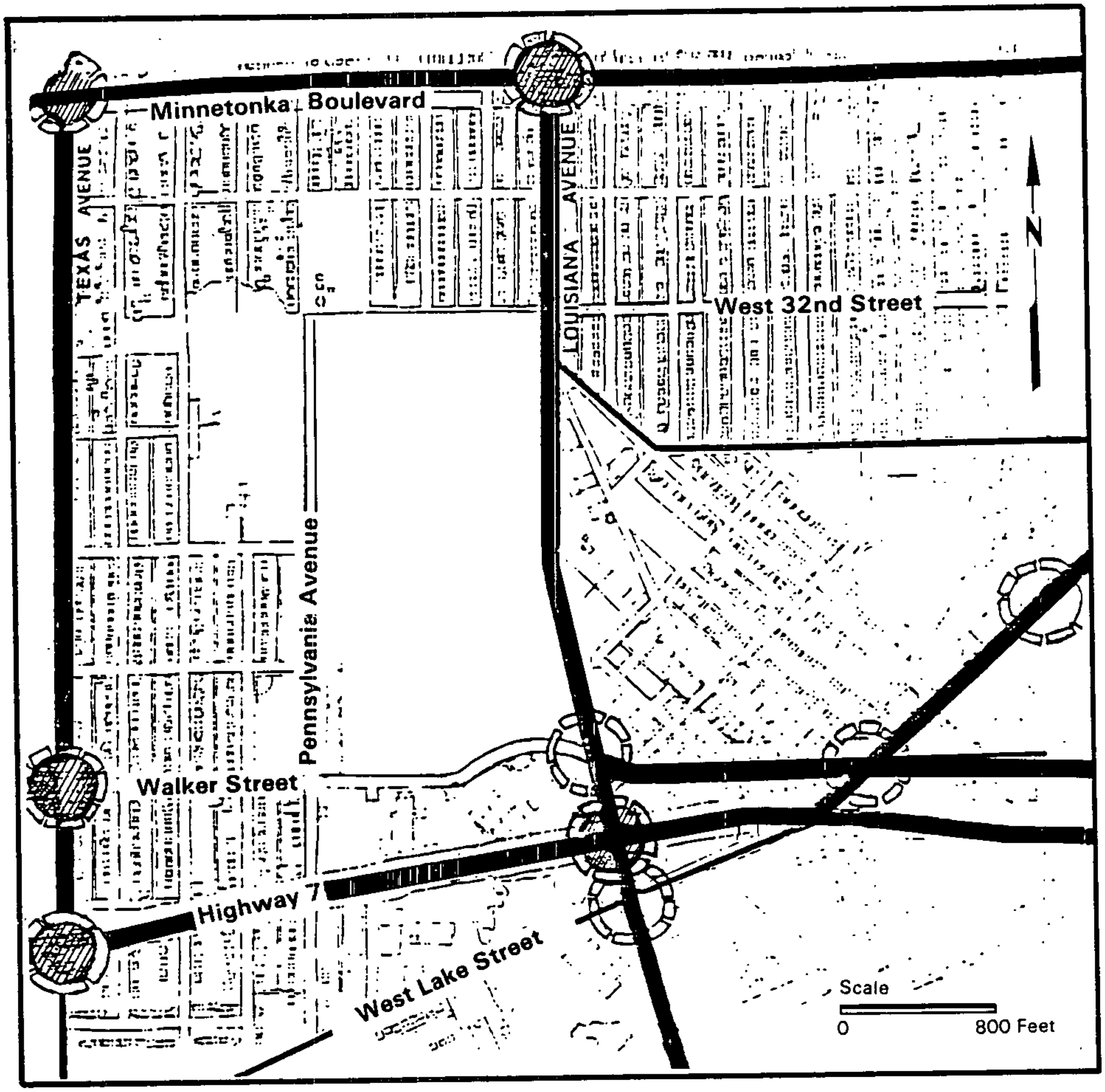


Figure A4-2 Map of Storm Sewer System



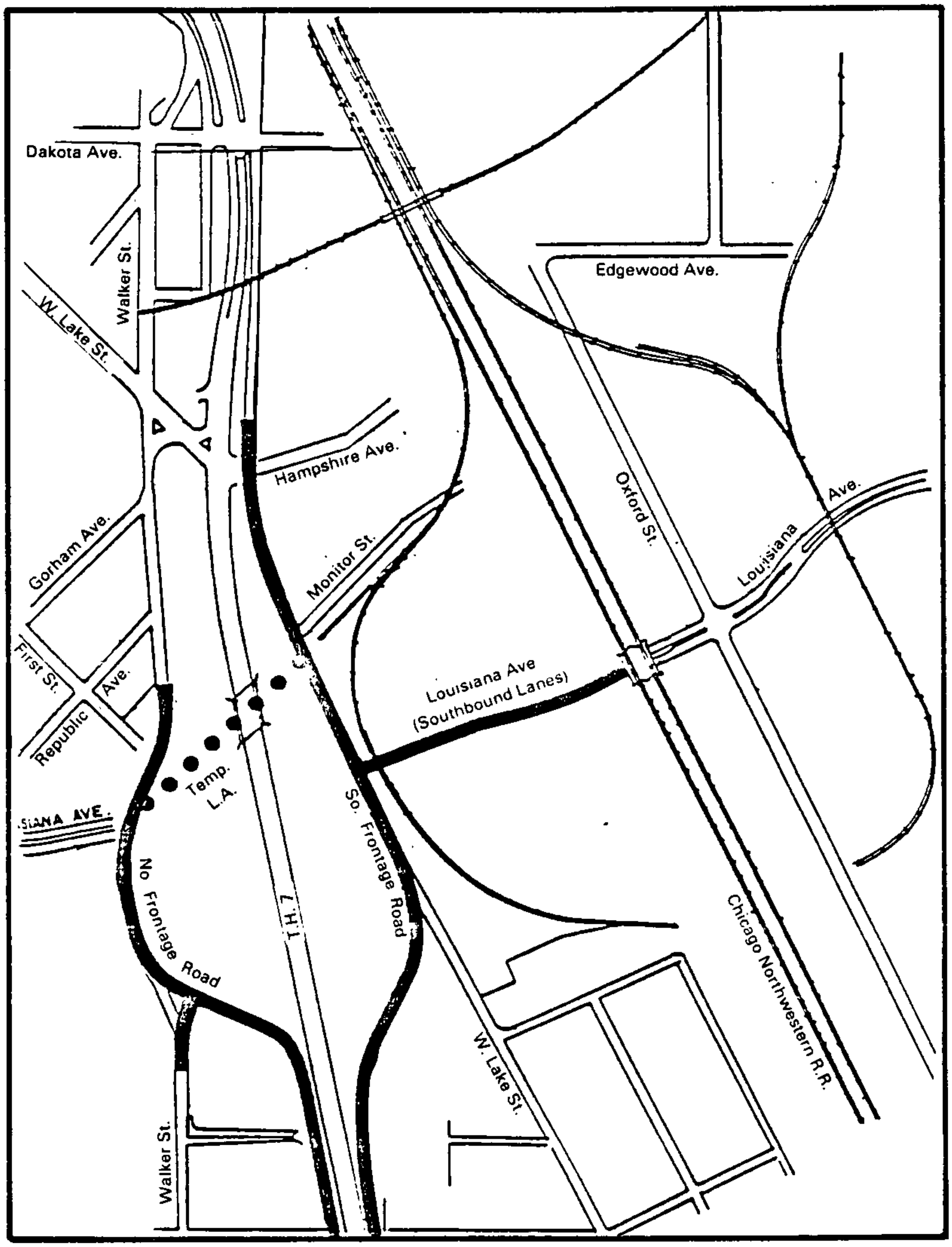
*Reproduced from National Biocentric Inc., 4/16/76, "Environmental Assessment Oak Park Village," Doc. No. 50002911-5002991.

EXPLANATION

- Major 4-Lane Streets
- Collector Streets
- Thru Traffic Intersections
- Signaled Intersections

Figure A4-3 Map of Planned Louisiana Avenue Extension

8303022



*Reporduced from City of SLP, "Proposed Louisiana Avenue Interim Solution," January 1980, Doc. No. 105045-105061.

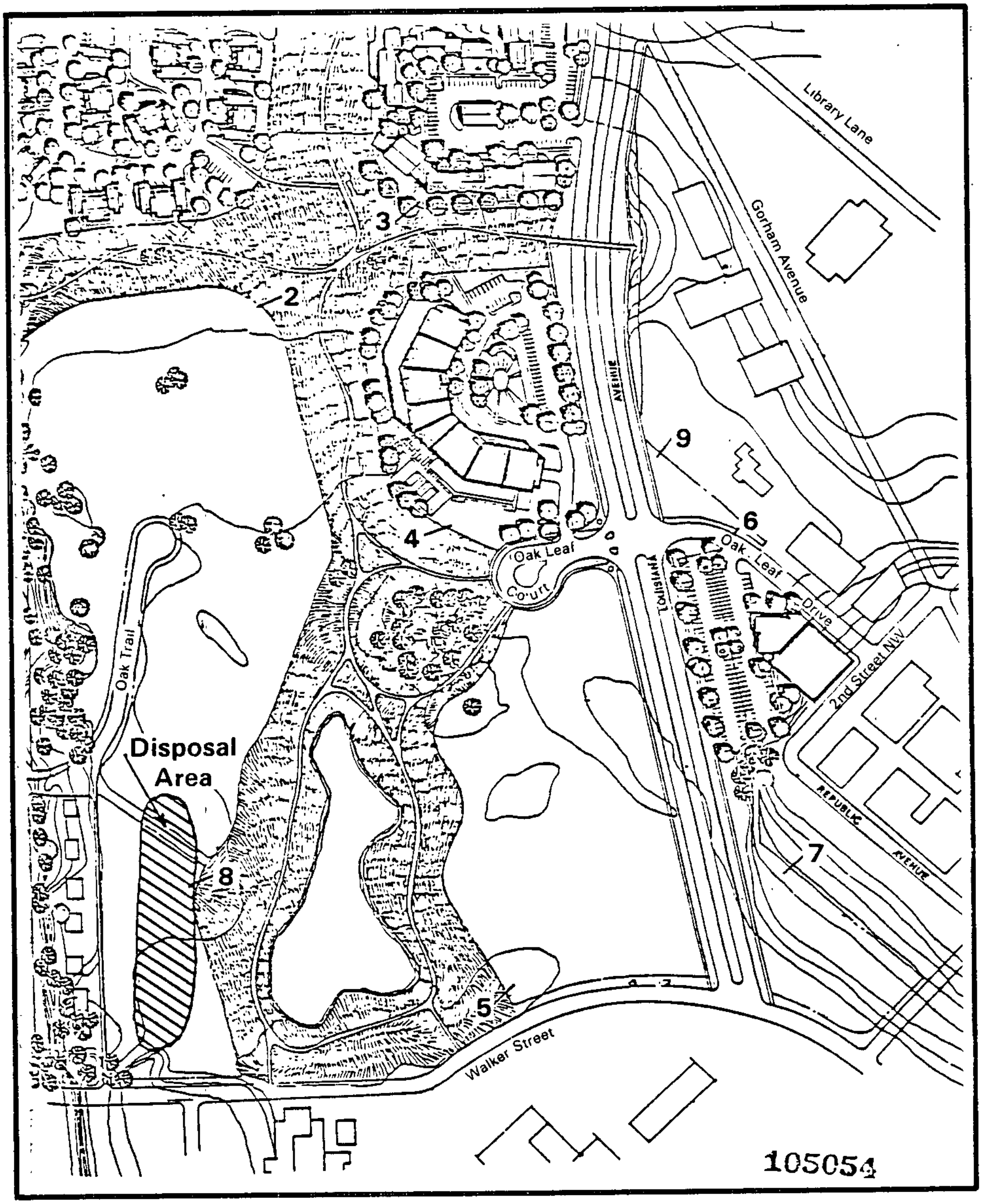
EXPLANATION

Construction as Planned

Temporary Construction

Figure A4-4 Map of Proposed Louisiana Avenue Extension Interim Solution

8303023



*Reproduced from City of SLP, "Proposed Louisiana Avenue Interim Solution," January 1980, Doc. No. 105045-105061.

Figure A4-5 Location of Contaminated Soil Stockpile

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Figure A4-6 Revised Oak Park Village Plan

Figure A4-7 Aerial Photograph of Site, November 26, 1982

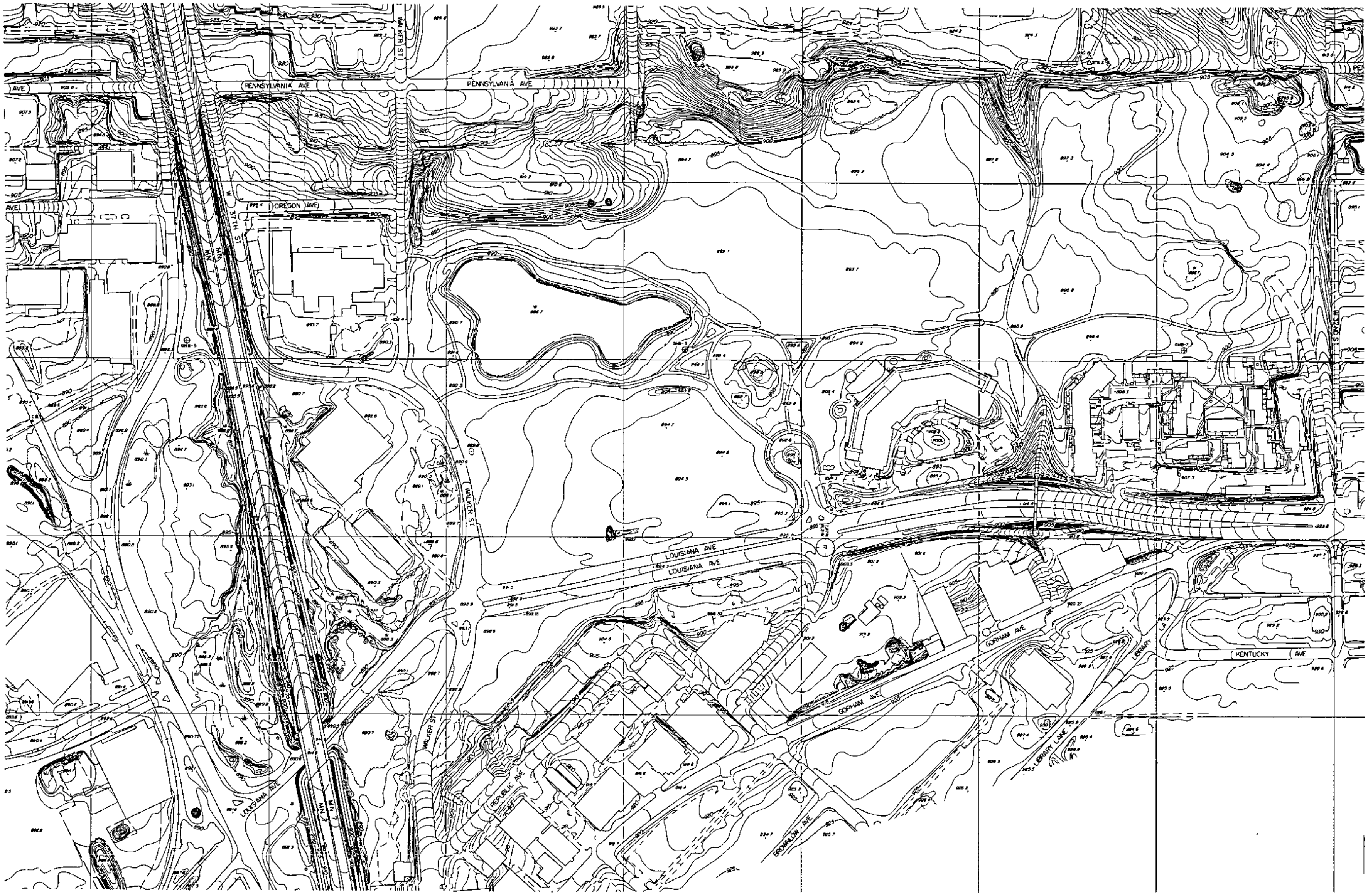

Figure A4-8 Topographic Map of Site, November 26, 1982

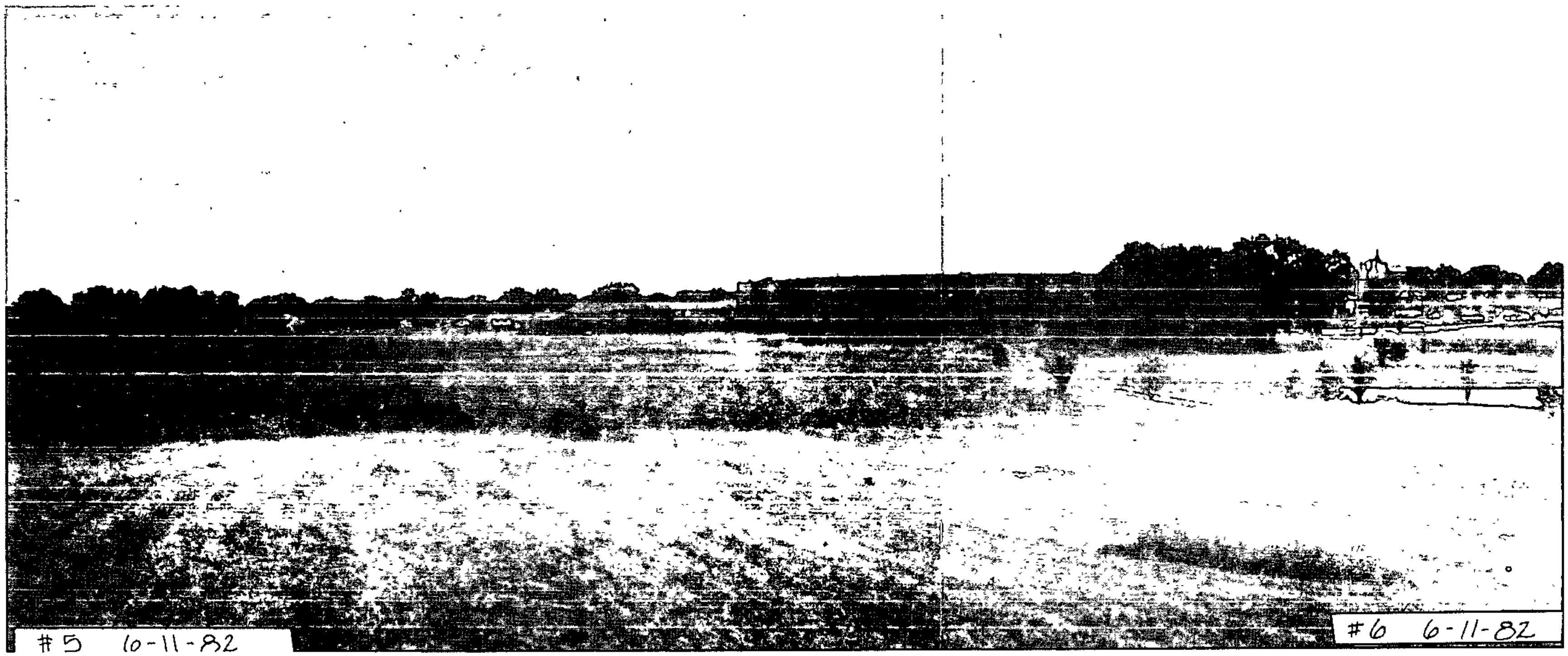

Figure A4-9 Current Photograph of Plant Site, View from Southwest Looking Northeast

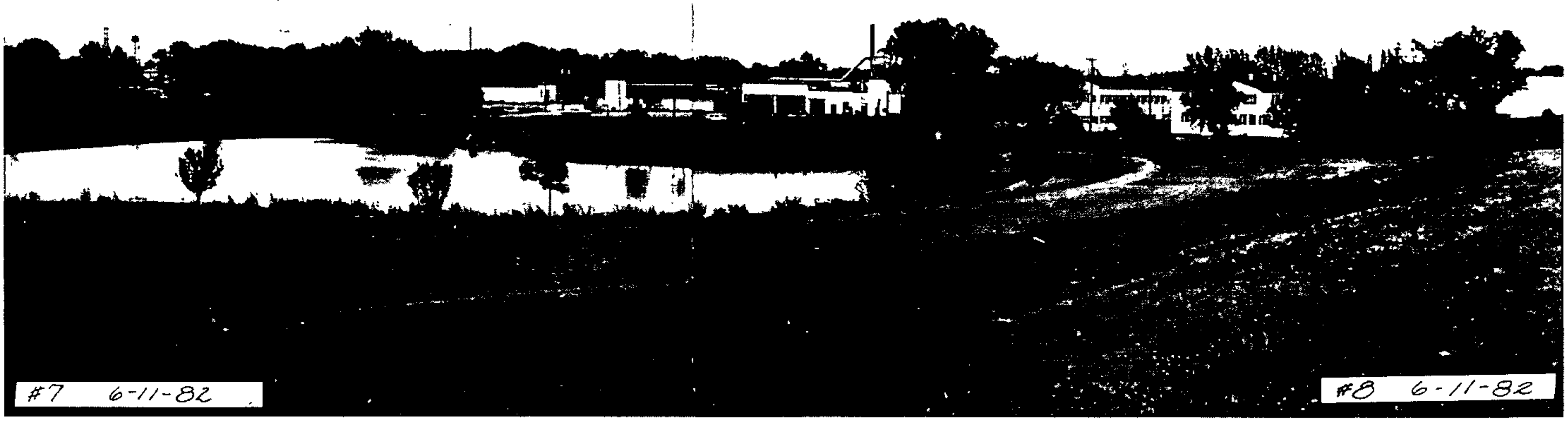

Figure A4-10   Current Photograph of Plant Site, View from North Looking South

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APPENDIX B

SITE CHARACTERIZATION

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B1. INTRODUCTION

In this appendix, the plant site and the adjacent bog are characterized in terms of distinguishing soil (naturally occurring surficial deposits and artificial fill) that contain concentrations of phenolics and benzene extractable hydrocarbons and surficial ground water (in the Drift-Platteville aquifer) that contain concentrations of polynuclear aromatic hydrocarbons (PAH) and related compounds that exceed estimated natural and anthropogenic (introduced by human activities) background concentrations.

B1.1 Objectives

The two objectives of this site characterization are to: (1) areally and vertically delimit the extent of and describe the nature of the occurrence of soil and surficial ground water contaminated by benzene extractable hydrocarbons, phenolics, and PAH that are present in concentrations above background concentrations; and (2) areally and vertically delimit the extent to which these phenolics and PAH have migrated from the plant site and the adjacent bog via the surficial deposits. The objectives are designed to facilitate assessment of the applicability and effectiveness of remedial action alternatives, and the potential for on-site contamination of soil and surficial ground water to be a source of contamination in the deep bedrock aquifers.

These objectives have been pursued by examining and synthesizing the findings of several investigations conducted at the site (Section B2). The background concentrations of indicators of contamination and the hydrogeologic framework in which the contamination occurs are derived from this synthesis of previous investigations. Examination of the spatial distribution of constituent concentrations that exceed background levels in the hydrogeologic framework has enabled delimitation of the contaminated soil and the extent of migration.

B1.2 Description of the Site

The plant site is a flat lying 80-acre parcel of land (Figure B1-1). It is bounded on the north by West 32nd Street, on the east by Gorham Avenue, Second Street Northwest and Republic Avenue, on the south by Walker Street, and on the west by Pennsylvania Avenue and Oak Hill Park.

None of the manufacturing facilities remains onsite. They were razed and removed in 1972. The northeast corner of the site is currently occupied by a condominium development. Louisiana Avenue has been extended through the eastern side of the site. It separates the eastern edge of the plant site that now supports several commercial buildings from the main part of the plant site. The southern half of the plant site has been graded and seeded and serves as a park. The City of St. Louis Park has constructed a lined runoff-collection pond within this park. This pond discharges via concrete culverts to a second lined pond approximately 1600 feet to the south. Discharge from the second lined pond is pumped to Minnehaha Creek. The northwest quarter of the plant site is undeveloped, and it is vegetated with a growth of weeds and grass. A footpath embankment rises from the west side of the lower end of this undeveloped portion of the site up to Oak Hill Park. Another footpath embankment has been constructed at the new route of Louisiana Avenue on the east side of the plant site in the middle of the condominium development.

The adjacent bog is an approximately 24-acre parcel of former wetland; less than one quarter of it remains unfilled. It is bounded on the north by Walker Street, on the east by the temporary route of Louisiana Avenue, on the south by Lake Street and on the west by former railroad embankments. Mill City Plywood and Mobile Marine Discount are situated in the predominantly filled section of the bog north of the Highway 7 embankment. Highway 7 cuts through the middle of the bog, and it is supported on a fill embankment that reaches approximately 25 feet in height at the bridge over the temporary route of Louisiana Avenue. The west half of the bog between Highway 7 and Lake Street has been filled, and the western most portion of the fill supports a new continuation of South Frontage Street to Lake Street.

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Figure B1-1 Map of Major Surface Features at the Site

Three small areas of open wetland remain in the bog area. One is located along the north side of Highway 7, another along the south side of the highway, and one patch along the south side of Walker Street.

The area of investigation includes the plant site and the bog. The surface of the plant site received contaminants as the result of various manufacturing processes and material handling operations which included coal-tar distillation and treatment of timbers with creosote. The bog received contaminants via runoff from the plant site. Surface-water runoff from that area was collected into a ditch that channelled flow under what is currently Walker Street. The bog was continuous until Highway 7 was constructed across the middle of it in the 1930's. A culvert under Highway 7 at the west end of the bog connected the two portions of the bog thereafter. Surface water in the bog discharged to Minnehaha Creek until the 1930's when the culvert under Lake Street collapsed and restricted the discharge route.

The plant site received contaminants directly from process and handling operations whereas the bog received runoff and wastewater from the plant site that was contaminated. These two areas are considered the primary receptors of contaminants that originated from manufacturing processes. These areas which constitute the site are characterized in the following sections in terms of the nature and extent of contamination found in the soil and ground water underlying these areas. Hereinafter, the plant site and the adjacent bog will be collectively referred to as the site. The characterization is based on interpretation of field and analytical work performed by the firms discussed in Section B2. No new data were acquired for this interpretation.

B2. DATA BASE

This section reviews the available sources of data that were used in compiling the interpretations presented in this appendix. The limitations imposed on the interpretations by these data are also discussed.

B2.1 Sources of Available Data

The project reports which describe subsurface conditions at the site that were used in the synthesis of this site characterization are listed in Table B2-1. A total of 17 projects are listed. These projects were undertaken between October, 1968 and June, 1979. Some of these projects were geotechnical investigations which described the soil in terms of suitability for foundations. Other projects were undertaken to investigate the nature of contamination of the soil underlying the site.

A total of 127 borings were drilled for these projects. Samples for chemical analysis were taken from only 56 of these borings. These borings produced 335 soil samples that were subjected to various chemical analyses. Of the 335 samples, 265 were analyzed for both phenolics and extractable hydrocarbons, 40 were analyzed for extractable hydrocarbons only, and 30 were analyzed for phenolics only. In addition, 25 samples were analyzed for specific PAH as well as for phenolics and/or extractable hydrocarbons.

Chemical analyses of 258 of the 335 soil analyses were carried out by Serco Laboratories, Inc. whose execution of protocols is well documented. The methods of collection in the field varied and are not well documented except for Barr Engineering Company (Barr) (1976 and 1977) investigations. Many different operators collected and initially handled the samples. The boring logs included in many of the reports that are supposed to correlate sampling interval, soil characteristics and genetic nomenclature (naming of soil types that indicates the origin of the deposit) are often of poor quality, incomplete and confusing. The highly variable format and quality

TABLE B2-1

SUMMARY OF SOIL BORING PROJECTS IN VICINITY OF SITE, 1968-1981

Map Code	Dates When Borings Were Drilled	Field Work Contractor	Client	No. of Borings	No. of Samples Taken for Chem. Analysis	Logs Available	Remarks	References
A	October 10, 1968	Soil Engineering Services, Inc.	Republic Creosoting Co.	2	0	Yes	Borings were 18 and 20 feet deep. Purpose was to determine soil conditions for hydraulic loader foundation.	Soil Engineering Services, Inc. 1968
B	September 1969	E.A. Hickok & Assoc.	Burdick Grain Co.	7	18	Yes	Borings were 13 to 18 feet deep. Soil was sampled every five feet. Boring number 5 is not located.	Hickok, 1969
C	Sept. 29 to Oct. 10, 1969	Soil Engineering Serv., Inc.	Republic Creosoting Co.,	23	0	Yes	Borings were 25 to 50 feet deep. Purpose of borings was to establish property value and evaluate foundation conditions.	Soil Engineering Services, Inc. 1969
D	August 4, 1970	Soil Engineering Serv., Inc.	Republic Creosoting Co.	6	0	Yes	Borings were 20 to 25 feet deep. Purpose was to evaluate soil and foundation conditions for a pipe line and separator unit.	Soil Engineering Services, Inc. 1970
E	April 23, 1971	Soil Engineering Serv., Inc.	City of St. Louis Park	6	31	Yes	Borings were 20 to 35 feet deep. Preconstruction soil evaluation for City sewer.	Soil Engineering Services, Inc. 1971

TABLE B2-1 (Continued)

Map Code	Dates When Borings Were Drilled	Field Work Contractor	Client	No. of Borings	No. of Samples Taken for Chem. Analysis	Logs Available	Remarks	References
F	March 1973	Orr, Schelen, Mayeron & Associates, Inc.	City of St. Louis Park	12	12	No	Purpose was to determine phenol concentration throughout the site. No logs accompanied the report and it may be that the samples were collected from the surface and no borings were actually drilled.	Orr, Schelen, Mayeron & Assoc. Inc., 1973
G	Feb. 11 to Feb. 25, 1974	Soil Exploration Co.	City of St. Louis Park	17	7	Yes	Borings were 22 to 68 1/2 feet deep. Samples were taken from three holes at approximately 20 foot intervals. Purpose was to evaluate soil and groundwater conditions for storm sewer system.	Soil Exploration Co., 1974
H	August 19, 1975	Soil Exploration Co.	Orr, Schelen, Mayeron & Assoc., Inc.	2	0	Yes	Both borings were 14 feet deep. Purpose was to determine soil conditions under the storm water retention pond on the site.	Soil Exploration Co., 1975
I	Nov. 1975 to Jan. 1976	Barr Engineering Co.	City of St. Louis Park	14	190	Yes	Borings were 45 to 75 feet deep. Samples were collected at five foot intervals where contamination was visible. Purpose was to define soil characteristics and measure amount of contamination.	Barr Engineering Co., 1976

TABLE B2-1 (Continued)

Map Code	Dates When Borings Were Drilled	Field Work Contractor	Client	No. of Borings	No. of Samples Taken for Chem. Analysis	Logs Available	Remarks	References
J	June 14-15, 1976	National Biocentric, Inc.	City of St. Louis Park	18	0	Yes	Borings were 10 to 19 feet deep. Purpose was to identify extent of visual contamination within the top 10 feet on northern portion of site.	City of St. Louis Park, 1976
K	August 13, 1976	National Biocentric, Inc.	City of St. Louis Park	4	9	Yes	Borings were 12 to 52 1/2 feet deep. Samples were taken at irregular depths. Purpose was to measure the amount of contamination.	National Biocentric, Inc, 1976
L	May 12-18, 1977	Soil Exploration Co.	Barr Engineering Co.	3	0	No	Borings are 70 1/2 to 108 1/2 feet deep. They were placed in order to better understand the geology of the area and piezometers were installed to give levels at the base of the drift.	Soil Exploration Co., 1977
M	March 10-22, 1978	Soil Exploration Co.	City of St. Louis Park	6	42	Yes	Borings were 69 1/2 to 77 feet deep. Samples were collected at five foot intervals. Purpose was to determine soil conditions for the proposed Louisiana Extension and collect samples for chemical analysis to measure contamination.	Soil Exploration Co. 1978

TABLE B2-1 (Continued)

Map Code	Dates When Borings Were Drilled	Field Work Contractor	Client	No. of Borings	No. of Samples Taken for Chem. Analysis	Logs Available	Remarks	References
N	October 9-19, 1978	Soil Exploration Co.	Serco Laboratories	4	26	Yes	Borings were 73 to 77 1/2 feet deep. Samples were taken at five foot intervals and every other one was analyzed. Purpose was to investigate soil conditions for proposed construction of Oak Park Plaza and a bowling alley.	Soil Exploration Co. 1978
O	October 19-24, 1978	Soil Exploration Co.	U.S. Geological Survey, Water Resources Division	2	0	Yes	Borings are 50 and 67 feet deep. Purpose was to install piezometers for monitoring water levels.	Soil Exploration Co., 1978
P	June 8, 1979	Braun Engineering & Testing	Environmental Research & Technology, Inc.	1	0	Yes	Boring is 50 1/2 feet deep. Samples were collected but were of inadequate size to analyze.	Braun Engineering Testing, 1979
O	1975 - 1981?	Contractor for U.S. Geological Survey	U.S. Geological Survey	?	?	No	Borings were used for installation of wells and piezometers for extracting water samples and measuring piezometric heads.	Ehrlich, 1982
			Totals	127	335			

NOTE: Only those borings that had logs adequate for reliable interpretation are located on Figure B4-1.

rendered correlation among the logs extremely difficult. None of the boring locations was surveyed. The locations of borings provided in the various reports are sketched on rough base maps. The compilation of well locations contained in Hult and Schoenberg (1981) gives the latitude and longitude of the wellhead down to the nearest second of arc. This best documented level of precision permits location of wells within a 75 foot by 100 foot error ellipse (Baier 1983). It is assumed that the soil borings were located on the sketch maps with no greater accuracy. The locations of the borings appearing on figures in following sections are based on sketch maps included in the various reports. These locations are approximate, and they may be in error in some cases on the order of 100 feet.

The inconsistencies in the reported boring and analytical data by the various investigators make selection of the site characterization data critical. This study has selected the boring-log data and analyses for phenolics and benzene extractable hydrocarbons that were prepared by Barr (1976 and 1977) as the basis for this site characterization. These data are the largest set of internally consistent information on physical soil description and soil contamination. These data represent concentrations that were measured seven years ago. The distribution and concentration of contaminants in the soil have probably changed due to physical, chemical, and biological attenuation and degradation. Data from other reports have been included where they could be reliably correlated with the detailed Barr data. The data that are not explicitly presented in this interpretation have, however, been thoroughly reviewed. From extensive knowledge of these data, it is known that these data do not contradict this interpretation.

Description of the nature and extent of surficial ground-water contamination is primarily based on the data prepared by Barr (1976 and 1977), Hult and Schoenberg (1981) and Ehrlich et al. (1982). The indicator parameters that have been selected for the interpretation of the contamination of surficial ground water are phenolics and PAH. One or both of these groups of compounds have been of major concern in the recent studies (Barr 1976 and 1977, Hult and Schoenberg 1981, Ehrlich et al. 1982), and both are constituents of coal tar although they are by no means peculiar to coal tar. Section B5.2 discusses in detail the selection of these indicator parameters and the quality of the data used in the interpretation of surficial ground-water contamination.

B2.2 Limitations of the Available Data

Barr sampled at five foot intervals vertically and at 17 locations over approximately 100 acres. For purposes of detailed interpretation of the complex surficial stratigraphy (configuration of soil types and classification of soil types by mode of deposition) and the nature and extent of the discontinuous contamination, the available data are inadequate. This is specifically the case for: 1) determining the thickness and lateral continuity of relatively thin fine-grained units such as the tills (unstratified, heterogeneous deposits laid down under a glacier) and lacustrine (lake) clay which to a great extent control the migration and local accumulation of the indicator compounds; and 2) determining the configuration of the subsurface accumulations of the indicator compounds and their defining concentrations.

These limitations of the data necessitate that any interpretation of the configuration of contaminated volumes of soil and the volume through which a measured concentration of an indicator compound can be projected be qualitatively described and schematically drawn on sections and maps. That is, any attempt at delimiting an area or volume of contaminated soil is done so schematically. The lines that delimit areas of a certain grain-size distribution or contaminated soil on the figures appearing in Sections B4 and B5 are schematic in the horizontal direction. The lens shape given to the areas only indicates that the grain-size distribution or indicator compound concentration cannot be interpolated between borings, and therefore the area of characteristic grain-size distribution or concentration is schematically terminated by the lens shaped area.

These limitations of the data and the complexity of the surficial deposits render contouring areas of equal indicator compound concentrations inaccurate and misleading. Examination of the spatial arrangement of concentrations of indicator compounds that are above background concentrations (Section B5) and their relation to the complex configuration of the surficial deposits which control the accumulation and paths of migration of these compounds reveals that elevated concentrations of indicator compounds occur in discontinuous bodies. The data are grossly insufficient to allow contouring of concentrations between any two adjacent discrete bodies of contamination and especially among the borings.

B2.3 Unavailable Data

The United States Geological Survey (USGS) and the Environmental Protection Agency (EPA) have been conducting investigations in the site area during and since publication of the latest private contractor report (Hickok 1981). Some of these data are reported by Hult and Schoenberg (1981) and Ehrlich et al. (1982), however, these publications do not present all of the public agencies' data acquired prior to publication. ERT has requested these data during the fall and winter of 1982. None has been received as of the publication of this report. Although these new data would refine the interpretations and conclusions presented in this appendix, the currently available data are believed adequate to characterize the site, evaluate its impact and assess remedial action options.

B3. SOIL CONTAMINATION CRITERIA

B3.1 Contamination Indicator Compounds

In this appendix, benzene extractable hydrocarbons and phenolics have been chosen to be indicators of soil contamination by coal-tar constituents. Hydrocarbon in this definition is not meant to be restricted to materials containing carbon and hydrogen, but rather it is meant to include all organic material so extracted. These parameters were used by Barr (1976 and 1977) to assess soil contamination. Although more recent work examined limited data on one type of PAH concentration in soil (not including bog deposits) (Hickock 1981), these data cannot be readily correlated with the much larger body of benzene extractable hydrocarbon and phenolic data. In addition there are no PAH data for the peat and organic silt underlying the bog which has been identified by others as the source of shallow ground-water contamination (Barr 1977, Hickock 1981 and Ehrlich et al. 1982).

B3.2 Background Concentrations of Indicator Compounds

A sample of soil is deemed contaminated if it contains concentrations of indicator compounds that exceed background concentrations typical for the type of soil from which the sample was taken. It is important to emphasize that all types of soil in the site area contain some background concentration of phenolics and benzene extractable hydrocarbons. These background concentrations originate from anthropogenic as well as natural sources. The anthropogenic sources result from off-site activities that were not in any manner related to the former manufacturing operations of RT&CC, and they also proportionately result from former RT&CC manufacturing operations. For example, anthropogenic sources of benzene extractable hydrocarbons background concentrations can result from combustion products of fossil fuels, spills of fossil fuels and bituminous concrete pavement. The steady manufacture, use and disposal of various common commercial products, especially in industrial/commercial areas, result in at least low level accumulations of chemicals that exceed naturally occurring background concentrations. This increase represents the anthropogenic contribution to background concentrations.

There are only two background soil samples taken from the site area that have been analyzed for benzene extractable hydrocarbons. A benzene extractable hydrocarbon concentration of 22,300 milligrams per kilogram was obtained from a sample described as peat that was taken in March, 1978 by Soil Exploration Company near the Westwood Townhouses which are located two miles northwest of the site. The other sample was described as "sandy soil", and it produced a benzene extractable hydrocarbon concentration of 200 milligrams per kilogram. The "sandy soil" location is given by Soil Exploration Co. as "Roseville Soil" and the collection date was November, 1978.

In light of the lack of definitive background data, estimates of background concentrations were formulated. These estimates were based upon comparing quantitative measurements of phenolics and benzene extractable hydrocarbons to qualitative descriptions of contamination in the soil samples for each soil type. Table B3-1 is a summary of the data base used in this evaluation, and Table B3-2 presents the estimated background levels.

Dependence upon the qualitative descriptions of certain soil samples as being visibly contaminated introduces an unknown and unavoidable level of subjectivity in this analysis. The data are considered sufficient, however, for identifying approximate background levels. It is important to note the highly differing background concentrations for the various soil types. For example, a benzene extractable hydrocarbon concentration of 20,000 milligrams per kilogram is a typical background concentration for the peat underlying the site, but it would indicate distinct contamination of the glacio-lacustrine clay which has a benzene extractable hydrocarbon background-concentration of 1,000 milligrams per kilogram.

These background concentrations are ultimately used as standards to which the concentrations of phenolics and benzene extractable hydrocarbons in the soil samples are compared. When the background concentrations are exceeded, the samples are interpreted to be contaminated.

TABLE B3-1

DATA BASE SUMMARY FOR SOIL QUALITY EVALUATION

	Artificial Fill		Bog Deposit		Glacio-Lacustrine Deposit		Middle Drift		Lower Drift	
Data Range[1]	Total No.[2]	No. Visibly Contam.[3]	Total No.	No. Visibly Contam.	Total No.	No. Visibly Contam.	Total No.	No. Visibly Contam.	Total No.	No. Visibly Contam.
$P \leq 0.2$	-	-	-	-	-	-	7	-	11	-
$0.2 < P \leq 0.5$	1	-	1	-	1	-	21	-	14	-
$0.5 < P \leq 1.0$	2	1	4	2	7	-	16	1	14	-
$1.0 < P \leq 2.0$	1	-	1	-	5	2	8	1	-	-
$2.0 < P \leq 5.0$	3	3	2	-	-	-	8	3	6	-
$5.0 < P \leq 10.0$	1	1	1	-	3	3	4	3	1	-
$10.0 < P \leq 20.0$	1	1	1	-	3	2	1	1	-	-
$20.0 < P \leq 50.0$	1	1	5	4	-	-	-	-	-	-
$P > 50$	-	-	6	5	-	-	-	-	-	-
$B \leq 50$	-	-	-	-	-	-	5	-	11	-
$50 < B \leq 100$	-	-	-	-	2	-	14	-	19	-
$100 < B \leq 200$	-	-	-	-	6	1	17	1	22	-
$200 < B \leq 500$	3	-	-	-	1	-	12	-	-	-
$500 < B \leq 1,000$	2	1	1	-	1	-	1	1	3	-
$1,000 < B \leq 2,000$	1	1	1	1	1	1	5	4	2	-
$2,000 < B \leq 5,000$	1	1	3	-	4	1	3	1	-	-
$5,000 < B \leq 10,000$	-	-	3	1	1	1	1	-	-	-
$10,000 < B \leq 15,000$	1	1	4	1	3	2	1	1	-	-
$15,000 < B \leq 20,000$	1	1	2	2	1	1	1	1	-	-
$20,000 < B \leq 25,000$	-	-	3	1	-	-	-	-	-	-
$B > 25,000$	2	2	6	6	-	-	-	-	-	-

(1) Data range in milligrams per kilogram for phenolics (P) and benzene extractable hydrocarbons(B).

(2) Total number of samples from Barr (1976) and Soil Exploration Company (1978).

(3) Number of samples described in Barr (1976) as visibly contaminated.

TABLE B3-2

ESTIMATED BACKGROUND CONCENTRATIONS OF PHENOLICS AND BENZENE EXTRACTABLE HYDROCARBONS

	Background Level Milligrams per Kilogram		Comments	
Soil Type	Phenolics	Benzene Extractable Hydrocarbons	Phenolics	Benzene Extractables Hydrocarbons
Fill	1	500 - 1,000	7 of 8 samples ≥1 milligram per kilogram described as visibly contaminated.	1 of 4 samples <1000 milligrams per kilogram described as contaminated. All 6 samples >1000 milligrams per kilogram described as contaminated.
Peat and Organic Silt (Bog Deposits)	10	10,000 - 25,000	2 of 9 samples <10 milligrams per kilogram described as contaminated. 9 of 12 samples >10 milligrams per kilogram described as contaminated.	2 of 8 samples <10,000 milligrams per kilogram described as contaminated. All six samples >25,000 milligrams per kilogram contaminated. One background sample at 22,300 milligrams per kilogram.
Clay & Silt (Glacio-Lacustrine Deposits)	5	1,000	2 of 13 samples <5 milligrams per kilogram said to be contaminated. 5 of 6 samples >5 milligrams per kilogram said to be contaminated.	1 of 7 samples less than 1,000 milligrams per kilogram described as contaminated. 7 of 10 samples >1,000 milligrams per kilogram said to be contaminated.
Sand & Gravel (Glacio-Fluvial Deposits)	1 - 2	1,000	1 of 44 samples <1 milligram per kilogram described as contaminated. 7 of 13 samples >2 milligrams per kilogram described as contaminated.	2 of 52 samples ≤1,000 milligrams per kilogram said to be contaminated. 7 of 11 samples >1,000 milligrams per kilogram described as contaminated.
Clay, Silt, Sand & Gravel (Undifferentiated Till and Glacio-Fluvial Deposits)	1	500	None called contaminated. 39 of 46 samples were ≤1.0 milligram per kilogram Maximum value was 7.8 milligrams per kilogram.	None called contaminated. 52 of 57 samples were ≤500 milligrams per kilogram Maximum value was 1,900 milligrams per kilogram.

B4. HYDROGEOLOGY

This section describes the artificial and natural unconsolidated deposits that underlie the site. In addition, the nature of ground-water flow beneath the site is described and correlated with the physical characteristics of the deposits. The descriptions contained in this section rely heavily on the figures which depict the spatial arrangement of the borings, cross-section alignments and the deposits.

The interpretations presented in this section are predominantly based on the data acquired by Barr (1976 and 1977). The surficial stratigraphy (genetic soil-type classification and configuration) has also been correlated with that of the USGS (Hult and Schoenberg 1981 and Ehrlich et al. 1982). Borings logged by the other contractors listed in Table B2-1 have been used in interpreting the subsurface conditions at the site, however, the logs of these borings are of distinctly poorer quality than Barr's logs. Therefore, correlation of the logs of the many contractors is incomplete and interpolation among the borings is also incomplete. The locations of the borings are shown on Figure B4-1.

B4.1 Surficial Deposits

The surficial deposits (artificial and natural) are described in the following sections in order of increasing depth. That is, the deposits are described in order of increasing age and in the order in which they would be encountered in an excavation. Figure B4-2 illustrates the surface configuration of the desposits which underlie the site, and Figure B4-3 illustrates the generalized configuration of the deposits with depth. Figures B4-4 through B4-7 (in the pocket at the back of the report) are cross-sections which illustrate the graphic logs of the borings on the section alignments and the interpretation of the surficial stratigraphy.

The graphic logs shown on Figures B4-4 and B4-5 illustrate the visual grain-size distribution descriptions given for the samples extracted from the borings and the approximate vertical extent through which a particular grain-size distribution occurs. The lateral extent of the lenses of distinct grain-size distributions are schematic.

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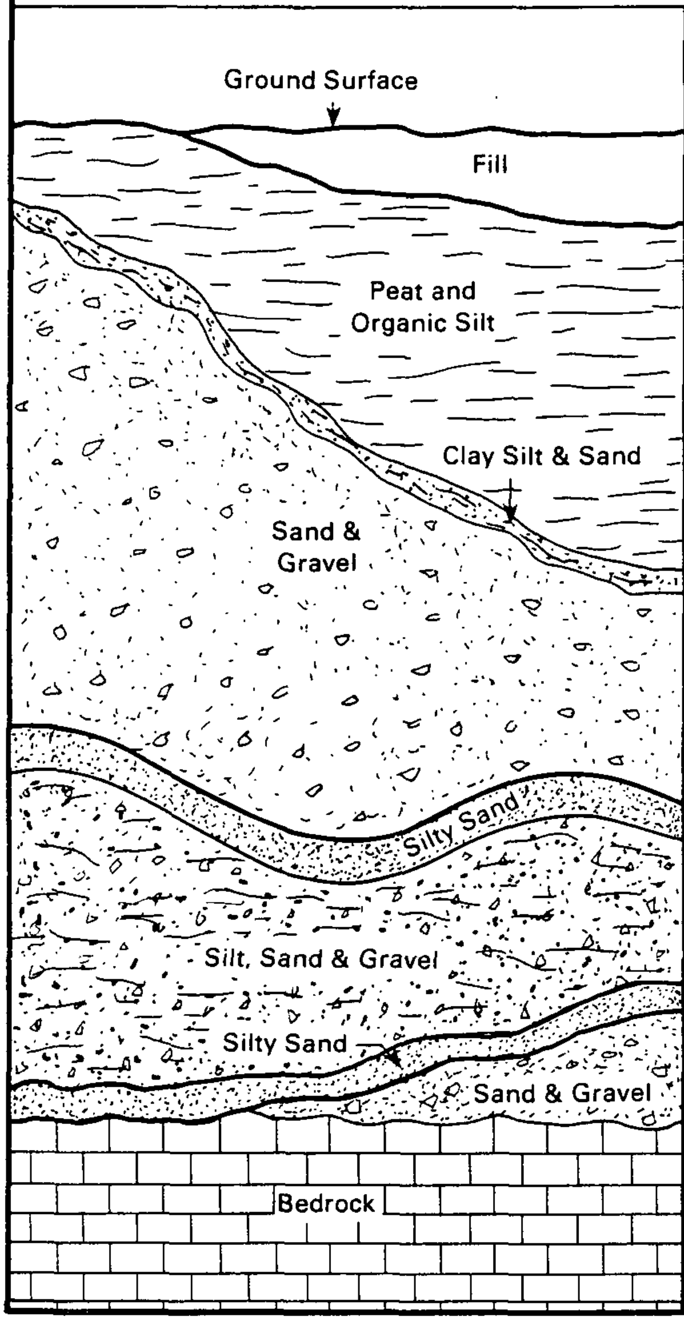
Unit Description



Drift	Unit Name	Thickness Range (ft)
	Artificial Fill	0-10
	Bog Deposit	0-33
	Glacio-Lacustrine Deposit	0-10
Middle Drift	Glacio-Fluvial Deposit	15-40
Lower Drift	Till	0-4
Lower Drift	Glacio-Fluvial Deposit	2-20
Lower Drift	Till	0-6
Lower Drift	Fluvial Deposit of Uncertain Origin	0-12
	Platteville Limestone	

Thickness, morphology and position of geologic units are based on boring information. This column represents a variety of conditions that may be encountered beneath the site; it may not depict all possible conditions.

Figure B4-3 Generalized Surficial Geologic Column

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They are, however, accurate in that they illustrate that the distinct grain-size distributions generally cannot be correlated among borings to form a stratum (a laterally persistent sheetlike or tabular deposit) of a particular grain-size distribution. Correlations among the borings are made on Figure B4-6 and B4-7 which illustrate the surficial stratigraphy. In these figures, geologic correlations are made to construct strata that are drawn based on distinct geologic modes of deposition. Grain-size distribution, however, can and does vary within a given genetic stratum.

B4.1.1 Artificial Fill

Artificial fill underlies most of the surface of the site. Its thickness ranges up to approximately 10 feet. The embankment for Highway 7, however, may be underlain by a thickness of fill on the order of 25 to 30 feet at the bridge over the temporary location of Louisiana Avenue. Fill has been placed on the site to produce construction grades above the seasonal high ground-water level, to control surface drainage and to improve bearing capacity. Visual descriptions of the fill indicate that it is typically a silty sand with varying amounts of gravel and occasional demolition debris.

An exposure of the fill in the sump constructed in the fall of 1982 at well W23 to collect fluid blown out of that well reveals two bands of black stained fill which resemble layers of pavement. The upper band is approximately six to eight inches thick and is overlain by four to six inches of topsoil and sod. The second band is approximately 12 inches thick and lies 18 inches below the surface. Similarly stained soil was also seen in the spoil pile next to the sump. This exposure is in the area of the former refinery. Barr (1976 and 1977) also reports "visible contamination" in the fill at nearby locations in the southern end of the plant site (borings I3, I4, I13 and I14).

B4.1.2 Bog Deposits

The bog deposits are composed of fibrous peat (probably sphagnum moss), muck (organic silt) and minor amounts of marl (organic silt and clay with a high proportion of precipitated calcium carbonate and/or shells). These deposits represent the natural ageing and filling process of a shallow lake. Much of the sediment in this deposit is of shallow lacustrine (lake) origin, however the bog deposit nomenclature was chosen to distinguish these highly organic deposits from the relatively inorganic underlying glacio-lacustrine (derived from a lake which formed at or near a glacier) deposits (Section B4.1.3). The muck and marl represent the earliest bottom deposits of the biologically active lake which succeeded the relatively biologically inactive glacio-lacustrine environment. As the lake became shallower and more bog like, increasing deposition of peat took place. These conditions are typical of the shallow lakes which abound in this area of Minnesota (Norvitch et al. 1973).

Bog deposits underlie much of the site in thicknesses of up to 33 feet in the southern most part of the site. These deposits pinch out (taper to extinction) to the east and west in the plant site (Figures B4-2 and B4-7). They are also absent in the central part of the plant site. Where the bog deposits occur below the plant site they are generally five to ten feet thick. Below the southern end of the site in what has been refered to as the "swamp", (Barr 1976 and 1977, Hickok 1981, Ehrlich et al. 1982) they are more typically on the order of 20 feet thick. These deposits are now exposed at the surface in only three small areas on either side of Highway 7 depicted by the wetland symbol on Figure B1-1. The remainder of the bog deposits have been covered with artificial fill.

B4.1.3 Glacio-Lacustrine Deposits

Visual descriptions of the grain-size distribution of the glacio-lacustrine deposits indicate that they are composed of inorganic silty clay. One mechanical grain-size analysis performed by

Barr (1976) produced these weight proportions: 35 per cent fine gravel (1 to 2 millimeters); 50 per cent coarse to fine sand (0.05 to 1 millimeter); 1 per cent silt (0.005 to 0.05 millimeter); and 14 per cent clay (less than 0.005 millimeter). This sample would more appropriately be called a clayey sand and fine gravel based on the weight proportion of its constituent grain-size distribution.

Although these sediments were deposited in the same lake as the bog deposits, glacial ice was in its latest wasting stage at the time of deposition and the climate was too cold to allow sufficient biologic activity to result in organic deposits.

The boring logs indicate that these deposits underlie the bog deposits wherever they occur. Boring E1 on Figures B4-5 and B4-7 shows a body of peat lying directly on glacio-fluvial deposits (Section B4.1.4), however, the sampling interval (probably five feet) could easily have missed the relatively thin glacio-lacustrine deposits. These deposits probably thin or pinch out locally, and they are breached by borings or possibly by excavations. They generally occur, however, in borings where they would be predicted to be. These deposits are interpreted to be a generally continuous bed of fine-grained (sand, silt and clay), inorganic sediment underlying the bog deposits. They are discontinuous under the site in the same pattern that was described for the bog deposits. The thickness of the glacio-lacustrine deposits ranges up to 10 feet in the northern end of the site. The bed is typically on the order of 0.5 to two feet thick.

B4.1.4 Glacio-Fluvial Deposits

The glacio-fluvial deposits are composed of varying distributions of coarse to fine sand and gravel, according to visual descriptions. The results of three mechanical grain-size analyses performed by Barr (1976) agree well with the visual descriptions. These deposits were laid down by running water that was discharged from the wasting glacial ice. They are generally void of fine grained sediment (silt and clay). This unit is also referred to as the Middle Drift aquifer. It underlies the entire site and is known to continue far off site (Hult and Schoenberg 1981, Ehrlich et al. 1982). Under the site it varies in thickness from 15 to 40 feet. At boring I12, it appears to be thickest, however, the five-foot sampling interval may have missed the upper till which would lead to an erroneously thick interpretation (Figure B4-6).

B4.1.5 Undifferentiated Glacio-Fluvial and Till Deposits

These deposits are composed of a complex configuration of glacio-fluvial (coarse to fine sand and gravel) and two till (unstratified, heterogeneous deposit laid down under a glacier) horizons composed of silt, clay and fine sand. The deposits are also collectively referred to as the lower drift aquifer. Two mechanical grain-size analyses performed by Barr (1976) on the glacio-fluvial sediment contained in this unit produced these weight proportions: 28 to 40 per cent gravel (greater than 1 millimeter); 50 to 72 per cent coarse to fine sand (0.05 to 1 millimeter); and one of the samples contained 10 per cent silt and clay (less than 0.05 millimeter). The visual descriptions of these glacio-fluvial deposits agree well with the results of the mechanical analyses. Barr (1976) also ran three mechanical analyses on samples of the lower till unit. The results of these analyses are: 28 to 47 per cent gravel; 33 to 54 per cent coarse to fine sand; and 16 to 33 per cent silt and clay. The results of the mechanical analyses indicate that the visual descriptions usually overestimated the relative amount of the silt and clay fraction.

Figure B4-3 schematically illustrates the four deposits that comprise this unit. Figures B4-6 and B4-7 illustrate these deposits as one collective unit. This was done because the data from the borings do not support a consistent differentiation of the discrete deposits of the unit over the distance between borings. This is especially true for the till units which are often very thin, on the order of one to five feet. The samples were taken at five foot intervals; therefore a thin, one to three foot thick deposit could easily be missed. In addition, the complex occurrence of lenses of various grain-size distributions could readily lead to an incorrect correlation. Barr (1977) presented plausible interpolations between borings for the till deposits. These interpolations, however, overemphasize the assumed continuity of the thin till deposits. It is more appropriate to assume that the till deposits are locally discontinuous. This interpretation is considered more appropriate than Barr's because the running water that deposited the upper and lower glacio-fluvial units was most likely capable of locally scouring away the till over which the water coursed.

The lower-most surficial deposit below the lower till and overlying the Platteville Limestone is visually described as being composed of coarse to fine sand with varying and minor amounts of gravel and silt. It is of fluvial origin, and probably is an interglacial stream deposit.

The undifferentiated basal unit as a whole varies in thickness from 10 to 30 feet under the site, and is, as a whole, continuous. The surface of the Platteville Limestone is weathered and its joints (fractures or breaks in the bedrock) are solutioned from ground water having dissolved faces of the joints in the limestone.

B4.2 Hydrology

Barr (1976 and 1977) and the USGS (as partially reported by Hult and Schoenberg 1981 and Erhlich et al. 1982) have installed a series of piezometers (small diameter wells) and wells to determine the nature of ground-water flow in the surficial sediments and to extract ground-water samples for chemical analysis. All of the piezometers and wells are screened in sediments below the glacio-lacustrine deposits including several in the Platteville Limestone and one in the St. Peter Sandstone. Measurements of the peizometric heads (the elevation to which ground water rises in a piezometer) indicate that ground water flows primarily laterally west to east and locally southeasterly through the Middle Drift, Lower Drift and Platteville aquifers in the site area (Figure B4-8). Downward hydraulic gradients have also been profiled below the site area (Barr 1976 and 1977, Hult and Schoenberg 1981 and Ehrlich et al. 1982). Barr (1977) integrated

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these vertical gradients into two hypothetical water-balance equations both of which indicated that at least half of the inflow to the middle drift aquifer migrates downward through the lower drift to the Platteville Limestone. The Platteville is considered to be confined at its base by the Glenwood Shale, and ground-water flow through the solutioned joints in the Platteville is assumed to be lateral in an easterly direction.

Hult and Schoenberg (1981) have mapped a tributary bedrock valley to the east of the site (Figure B4-8). The head of this valley lies 3,000 ft to the east of the bog. In this valley the Platteville and Glenwood bedrock units have been eroded and the St. Peter Sandstone is in direct contact with the glacial sediments. Hult and Schoenberg (1981), Ehrlich et al. (1982) and Barr (1977) have indicated that a steep, downward, vertical hydraulic gradient exists over and into this valley. This gradient is believed to carry a portion of the ground-water discharge from the site area down into the St. Peter Sandstone. Ground-water modeling done for this study indicates that approximately 30 per cent of the ground-water flow over the bedrock valley flows down from the Drift-Platteville aquifer into the underlying bedrock via the valley (Appendix E). The remainder of the discharge continues flowing easterly, ultimately to the Mississippi River (Norvitch et al. 1973). Ground water that passes below the southern part of the site area would flow east and some would tend to flow downward into the bedrock valley. Ground water that flows below the plant site would tend to flow in a similar manner, except that it would not likely be perturbed far enough to the south by the vertical gradient at the head of the valley to flow into it. Appendix E discusses in detail the development of a water balance which supports this general ground-water flow description.

Barr's (1977) water balances indicate an overall downward flow trend for ground water as it flows beneath the site. According to Barr's hypothesis, ground water entering the site via the Middle Drift from upgradient sources to the west would tend to descend from the Middle Drift through the upper till, lower till and Lower Drift into the Platteville Limestone in which it would flow in an easterly direction. Barr does not discuss in detail the portion of the precipitation or surface run-off flow onto the site. In addition, Barr does not fully integrate its water-balance scenarios with the nature and configuration of the deposits under the site. Also, the nature of the solutioned joints in the Platteville, which are the conduits of ground-water flow, is poorly known. For example, it is not known whether the joints are open, locally filled with sediment, or generally what the specific hydraulic characteristics of the Platteville are in the site area. Barr's hydrological interpretations are based on simplifying assumptions that do not take into account hydrogeological constraints. Barr's assumptions conflict with Hult and Schoenburg's (1981) vertical head measurements in the Middle Drift which indicate that flow is primarily lateral to the east. The interpretations of the USGS as presented in Hult and Schoenberg (1981) and Ehrlich et al. (1982) are believed more representative of the actual hydrology since they are based on a far larger set of data and the most recent data.

Erhlich et al. (1982) have mapped a generalized ground-water flow pattern through the surficial deposits that underlie the site that trends due east (Figure B4-8). Erhlich et al. (1982) also acknowledge distinct vertical gradients directed downward at the site. They describe the ground water that flows through the Middle Drift as discharging laterally to the east and southeast and vertically into the basal surficial unit. It is further stated that piezometric head differences between the top and bottom of the basal surficial unit indicate discharge through the basal unit into the Platteville Limestone.

Three nearly continuous fine-grained deposits underlie the site: one glacio-lacustrine, and two till deposits. The permeabilities of these deposits have been estimated to be on the order of 10^{-6} to 10^{-4} centimeters per second (Barr 1977). These low permeabilities are sufficient to confine ground-water flow beneath the site to predominantly horizontal easterly to southeasterly directed flow paths. In addition, the lenticular or tabular configuration of these fine-grained deposits would preferentially impede vertical flow and promote horizontal flow. Ehrlich et al. (1982) cite steepened vertical gradients resulting from the relatively low vertical permeability of these units. This indicates impedence of vertical flow.

The nature of ground-water flow through the surficial sediments in the site area is complex, and the data available for preparation of this interpretation are not sufficient to support a definitive explanation. The data, however, do indicate that the complex surficial geology produces complex discrete flow paths. This information is also sufficient to model the general path of ground-water borne contaminants emanating from the site. Ground water in the surficial deposits flows through the upper and lower drift in an easterly to southeasterly direction with a marked downward component, and there is some vertical leakage down through the surficial deposits into the Platteville. In addition, the Drift and Platteville are hydraulically linked (Appendix E).

Ground-water flow through the bog and glacio-lacustrine deposits has not been addressed by most of the previous investigations (Barr 1976 and 1977; Hickok 1981; Ehrlich et al. 1982). Also, there is no hydrologic information on these deposits. One mechanical grain-size analysis (Barr 1976) identifies a sample of glacio-lacturine sediment as a clayey coarse to fine sand and fine gravel. Its permeability is estimated to be 10^{-5} centimeters per second (Barr 1977). The ground-water saturated organic silt, peat and marl which compose the bog deposits are estimated to have a permeability within an order of magnitude (ten times) greater than the underlying glacio-lacustrine deposits, or approximately 10^{-4} centimeters per second. The configuration of the bog deposits is one of a saturated lens of organic sediments underlain by a saturated, continuous deposit of fine-grained, inorganic sediment that is distinctly less permeable.

Water enters the bog deposits via percolation through the overlying fill, run-off and direct precipitation into the exposed bog areas. Ground water in the bog discharges primarily via surface-water courses (man-made culverts), and by means of slow leakage through the glacio-lacustrine deposits to the Middle Drift. Vertical flow through the glacio-lacustrine deposits has most likely been augmented by the numerous borings that have punctured these deposits. This has probably also happened in the case of puncturing the low permeability till deposits. There is little information on how or whether the many wells and borings were sealed or grouted after completion of exploration or well installation. Puncturing the lenses and relatively continuous strata of fine-grained sediments produces conduits through which ground water can flow relatively rapidly in response to the documented downward vertical gradient. This effect will also increase the rate of contaminant movement in response to the vertical gradient and the greater density of some of the indicator compounds, and thus create a more complex distribution of contaminants.

Hult and Schoenberg (1981) describe the hydrology of the bog prior to cessation of wastewater input from the plant. Prior to the collapse of the bog-outlet culvert under Lake St. and the construction of Highway 7 in the mid-1930's, input wastewater and runoff could steadily flow off to the south to Minnehaha Creek. After this constriction of surface-water outflow occurred, the inflow to the bog area exceeded the outflow, infiltration and evapotranspiration rates which resulted in mounding of the water table and accelerated movement of contaminants into the underlying sediments. This condition has been fully mitigated by cessation of wastewater input since the plant closed in 1972, by the containment of runoff to lined ponds and storm sewers and by filling most of the bog. The runoff input to the bog is probably lower now than it has ever been since the runoff retention ponds and storm sewers were constructed. Therefore, recharge to the underlying sediments and attendent contaminant migration is probably also at its lowest rate since discharge of wastewater was initiated.

In summary, the detailed patterns of ground-water flow in response to the complex surficial stratigraphy are poorly understood. The general pattern of ground-water flow, however, can be modeled. Ground water flows beneath the site area in an easterly direction and locally responds to downward hydraulic gradients. The ground-water system of the bog deposits is like a surface-water system that primarily discharges down the surface hydraulic gradient and leaks slowly downward through the confining glacio-lacustrine deposits in response to the vertical hydraulic gradient.

B5. EXTENT OF CONTAMINATION

This section describes the areal and vertical extent of soil (including all unconsolidated natural deposits and artificial fill above bedrock) that contains concentrations of contaminants that exceed estimated background concentrations. The discussions in the following subsections rely heavily on the spatial relationships depicted in Figures B5-1 through B5-5. Figures B5-2 through B5-5 are in the pocket at the end of the report. In addition, this section describes the nature and extent of contamination in the surficial ground water.

Figure B5-1 illustrates the areal subdivision of the site based on the nature and level of contamination. Figures B5-2 and B5-3 illustrate the spatial relationship among the total concentrations of benzene extractable hydrocarbons and phenolics, the depth of the samples and the type of soil from which the sample was taken. These concentrations and soil descriptions were taken from Barr (1976 and 1977) as discussed in Section B2. Figures B5-4 and B5-5 depict those areas in which the total concentration of benzene extractable hydrocarbons and/or phenolics exceed the estimated background concentrations listed on Table B3-2. The numbers listed on either side of the boring trace indicate the multiple above the estimated background concentration for the soil type from which the sample was taken. If no multiple number appears on Figure B5-4 or B5-5 at a location for which a total concentration is reported on Figure B5-2 or B5-3, the concentration is within the background concentration range for the soil type from which the sample was taken. Reference to Table B3-2 and the discussion of contamination criteria in Section B3 is critical to putting the degree of contamination (multiple above estimated background concentration) in context with the total concentration. This is critical because the background concentrations vary through several orders of magnitude (factors of 10) based on the type of deposit on which the analysis was performed. For example, a benzene extractable hydrocarbon concentration of 20,000 milligrams per kilogram is a typical background concentration for the bog deposits underlying the site, but

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it would indicate distinct contamination of the glacio-lacustrine clay which has a benzene extractable hydrocarbon background-concentration of 1,000 milligrams per kilogram.

The cross sections were limited to the southern half of the plant site and the bog, because these two areas, contain the preponderant amount of contaminated soil.

B5.1 Areal and Vertical Extent of Soil Contamination

The original surface of the entire plant site has most likely received some amount of contaminants. For example, small amounts of contaminants have been received from splinters of treated wood, precipitation wash-off from stockpiled treated wood as well as from processes and accidents. The site has been divided into three areas based on the nature and the relative level of contamination (Figure B5-1). These areas are the northern half of the plant site, the southern half of the plant site and the predominantly filled bog to the south of the plant site.

B5.1.1 Northern Half of the Plant Site

This subdivision of the site was used primarily for stockpiling untreated and creosote-treated lumber. Minor amounts of coal-tar related material were applied to portions of this subdivision to stabilize or bind the surface soil. This site improvement was undertaken to increase the trafficability of this area. Seepage from stockpiled products and stabilized surface areas are the two most likely sources of contamination. These sources would produce contamination at very low levels.

Analytical data from soil samples taken from this subdivision of the site reveal concentrations of benzene extractable hydrocarbons and phenolics that are within estimated background concentrations. Two samples produced indicator compound concentrations that are elevated above background concentrations. These two samples (one each from borings I1 and I2) were taken from depths of less than 10 feet, and produced benzene extractable hydrocarbon concentrations that are 2 to

3 times the estimated background concentration of 1000 milligrams per kilogram or 2200 and 3050 milligrams per kilogram, although neither sample was described as odorous or visibly contaminated. Both of these samples were taken from the glacio-lacustrine deposits. In each boring, the soil above and below the glacio-lacustrine deposits contain benzene extractable hydrocarbons and phenolic concentrations that are within background concentrations. The fine grained glacio-lacustrine deposits have probably retarded the low level flux of benzene extractable hydrocarbons which were transported via percolating ground water from the original surface which in turn had received contaminants. This flux retardation manifests itself in the form of relatively elevated benzene extractable hydrocarbon concentrations. Since the surface of this subdivision has been extensively excavated, graded and developed, the source of the low level flux of benzene extractable hydrocarbons has been largely removed.

The eastern half of this subdivision has been intensely developed. This area now supports a formally landscaped condominium complex. A full array of buried utilities has been installed. During construction of the condominiums and installation of the buried utilities excavated soil was stockpiled in the southwest corner of the plant site. This excavation and stockpiling was done because it was believed that the excavated soil contained potentially harmful concentrations of contaminants (National Biocentrics 1976). The degree of contamination of this stockpiled soil is, however, not known.

These characteristics indicate that this subdivision has minimal potential of being a generator of contaminants that could enter the surficial ground-water and migrate off site. For this reason, the northern half of the plant site is not considered in the following discussions that describe the on-site source of surficial ground-water contamination on-site and downgradient from the site.

B5.1.2 Southern Half of the Plant Site

The southern half of the plant site supported the major process facilities. Among these facilities were the coal-tar refinery (located near borings I13 and I14) and the wood-treatment process equipment (located near borings I4 and I5).

Examination of the pattern of areas of contaminant concentrations that exceed background concentrations on Figures B5-3 and B5-5 indicate that the near-surface artificial fill underlying this subdivision was contaminated by benzene extractable hydrocarbons which were found in concentrations (1,020 to 44,400 milligrams per kilogram) from 1.4 to 59 times the background concentration (750 milligrams per kilogram) and by phenolics which were found in concentrations (1.2 to 26.3 milligrams per kilogram) from 1.2 to 26 times the background concentration (1 milligram per kilogram). Only the fill deposit manifests a pattern of general and areally persistent contamination. The contamination of the underlying natural deposits instead shows a discontinuous pattern of discrete volumes of contamination. Two of these discrete zones appear to be directly associated with a geologic control. Both of these zones are in boring I6; one is at elevation 830 feet, and the other is at elevation 850 feet (approximately 60 ft and 40 ft deep, respectively) (Figure B5-5). The upper zone contains phenolics (1.4 milligrams per kilogram) at a level of 1.4 times background (1.0 milligram per kilogram) within a lens of fine-grained sediment. The lower zone contains benzene extractable hydrocarbons (1050 milligrams per kilogram) at twice the background level (500 milligrams per kilogram), and it rests on top of a lens of fine-grained sediment. In borings I14 and I4 there are discrete zones of contamination that do not appear to be related to a geologic control (Figure B5-5). It is however, distinctly possible that the five foot sampling interval was too widely spaced to detect thin, lenses of fine-grained sediment that appear to generally control the location of these discrete zones of contamination.

The depth of continuous contamination from the top of the first sampling interval (generally two feet below the surface) varies from four feet at boring I6 to 18 feet at borings I4 and I5. Boring I4 contains the largest concentrations of benzene extractable hydrocarbons and phenolics in this subdivision. Through the upper 13 feet of the sampled portion of the boring, benzene extractable hydrocarbon concentrations range from 3.2 to 59 times the background concentration; phenolics similarly vary from 4.5 to 26 times the background concentrations. It is illustrated on Figures B5-3 and B5-4 that: the sampled portion of the artificial fill at I4 from 3 to 7 feet contains from 28,600 to 44,400 milligrams per kilogram benzene extractable hydrocarbons or from 38 to 59 times the background concentration of 750 milligrams per kilogram, and from 15.1 to 26.3 milligrams per kilogram phenolics or from 15 to 26 times the background concentration of 1 milligram per kilogram; bog deposits from 7 to 13 feet contain from 63,600 to 188,000 milligrams per kilogram benzene extractable hydrocarbons or from 3.2 to 9.4 times the background concentration of 20,000 milligrams per kilogram, and from 45 to 209 milligrams per kilogram phenolics or 4.5 to 21 times the background concentration of 10 milligrams per kilogram; the sampled portion of the glacio-lacustrine deposits from 13 to 15 feet contains 14,000 milligrams per kilogram benzene extractable hydrocarbons and 14.9 milligrams per kilogram phenolics; and the sampled portion of the glacio-fluvial deposits from 15 to 18 feet contains 640 to milligrams per kilogram benzene extractable hydrocarbons and 2.6 to milligrams per kilogram phenolics.

Within the depth of continuous contamination, there is a general, although not steady, decrease in level of contamination with depth. Boring I6 is peculiar in that distinct contamination is found to such great depth. Measurable levels of contamination are found in two discrete zones within the lower drift. One zone contains low levels of benzene extractable hydrocarbons (1,050 milligrams per kilogram) (two times background concentration of 500 milligrams per kilogram), and it lies between two thin beds of fine-grained sediment (Figure B5-5). The lowest zone which lies at the contact with the Platteville Limestone contains benzene extractable hydrocarbons at levels of 1,770 milligrams per kilogram or 3.5 times background (500 milligrams per kilogram), and phenolics contamination at 5.1 milligrams per kilogram or five times the background concentration (1 milligram per kilogram). The depth at which contamination is found, and the fact that the overlying sediments are only lightly and locally contaminated, together suggest that the manifestation of contamination at such a depth is the result of lateral and vertical migration. The source of this deep contamination is probably located near the surface and up the hydraulic gradient from boring I6. This boring is located downgradient from the former wood-treating area at which boring I4 was drilled. Boring I4 yielded samples containing the highest level of contaminant concentrations on site.

The southern half of the plant site is generally contaminated from the original surface (which has been regraded and seeded) to a depth of about 10 feet. The level of contamination appears to be greatest in the former wood-treatment areas as seen in the analytical results from boring I4; and in the area of the former coal-tar refinery as seen in the exposures in the sump at well W23 and in boring I14. The discrete bodies of contamination in the Middle and Lower Drift indicate that contaminants have migrated laterally and downward from the areas in which they were received at the surface.

B5.1.3 Bog

The bog to the south of the plant site is bounded on the north by Walker Street, on the south by Lake Street, on the east by the temporary Louisiana Avenue right-of-way and on the west by an unused railroad grade that lies to the west of the recently relocated South Frontage Street. This sub-division of the site received contaminants from surface run-off and wastewater that was discharged via an open channel and culvert under Walker Street. Prior to the mid-1930's when Highway 7 was constructed, the bog was continuous. The two halves of the the bog were thereafter connected via a culvert under Highway 7. A culvert under Lake Street conducted discharge from the bog to the low lying areas to south of Lake Street until it collapsed in 1932. Thereafter, surface discharge from the bog area was restricted, and infiltration to underlying deposits increased. Although runoff control and wastewater discharge practices steadily improved over the history of the plant operation, the bog served as a sink for some of the plant effluent throughout the plant's operational history (Appendix A).

The analytical data reveal that the peat and organic silt underlying the bog are generally contaminated from the original upper surface through a depth of 10 feet. The bog deposits are not, however, everywhere contaminated. In fact, at boring I9 (Figure B5-4) only the sampled portion of the uppermost five feet of a total of 27 feet of peat and organic silt is measurably contaminated. The level of phenolics contamination (14.8 milligrams per kilogram) at this point is very low, 1.5 times background level of 10 milligrams per kilogram. The concentration of benzene extractable hydrocarbons (13,000 milligrams per kilogram) is within the background level of 20,000 milligrams per kilogram.

It is not known whether the peat and organic silt were excavated prior to constructing Highway 7 and the buildings which house Mill City Plywood and Mobile Discount Marine. Since these structures were built well after runoff and wastewater were being regularly discharged to the bog, it is assumed that any peat and organic silt that were originally near the surface of the bog and are now beneath these structures are probably contaminated.

The eastern and western limits of the contaminated near surface sediments in Section B-B' are schematically drawn based on the phenolic concentrations obtained by Hickok (1969). In boring B2 a phenolic concentration of 0.021 parts per million is reported for the "black organic clay" which in this interpretation is correlated with the bog deposits. In addition, a phenolic concentration of 0.014 parts per million is reported for the same deposit in boring B3. Assuming that these concentrations were calculated on a mass basis, they are within the background concentration of 1 milligram per kilogram. Note that 1 part per million is equivalent to 1 milligram per kilogram.

The western limit of the contaminated original surface of the bog could not be determined in Section A-A'. There are no analytical or observational data available for this area although many borings were drilled for the storm sewer and runoff holding ponds by Soil Exploration Co. (1974). It is assumed that if any contamination is present in this area it is of a low level since the gravity line which connects the runoff collection ponds was constructed through this area, the area has been filled and South Frontage St. has been extended out over the new fill. The eastern limit was drawn between boring I11 and the railroad embankment (now the temporary right-of-way of Louisiana Avenue). On several maps dated 1898 the railroad embankments are shown. These embankments were in place at least 18 years prior to initiation of processes at the plant.

The south end of Section D-D' crosses through the same area of deficient information as the west end of section A-A'. The southern limit of the original bog-surface contamination could not be drawn. It is, however, known that ground-water samples from well W8 and soil samples from boring I7 are within background quality. Lake Street is taken as a reasonable estimate of the southern extent of original bog-surface contamination.

In a manner similar to that described for the northern end of the plant site, there are discrete zones of contamination at depth below the generally contaminated original bog surface. In most instances, these discrete zones of contamination are associated with lenses or strata of fine-grained sediments. Contamination is discernable above, within and immediately below these lenses of fine-grained sediments. The manner of occurrence of these zones of contamination again suggests that the fine-grained sediments not only control the flow of ground water, but they also control the accumulation and rate of migration of contaminants.

B5.1.4 General Pattern of Soil Contamination

The regular occurrence of discrete bodies of contaminated soil throughout the surficial deposits underlying the site at depth below and separate from the overlying areas which received the initial inflow of contaminants strongly implies that the deposits of fine-grained sediments (i.e., the glacio-lacustrine deposits and the till deposits) are not homogeneously persistent across the site. That is, they are thin or absent in areas, and they vary in grain-size distribution and permeability to ground-water flow and contaminant migration.

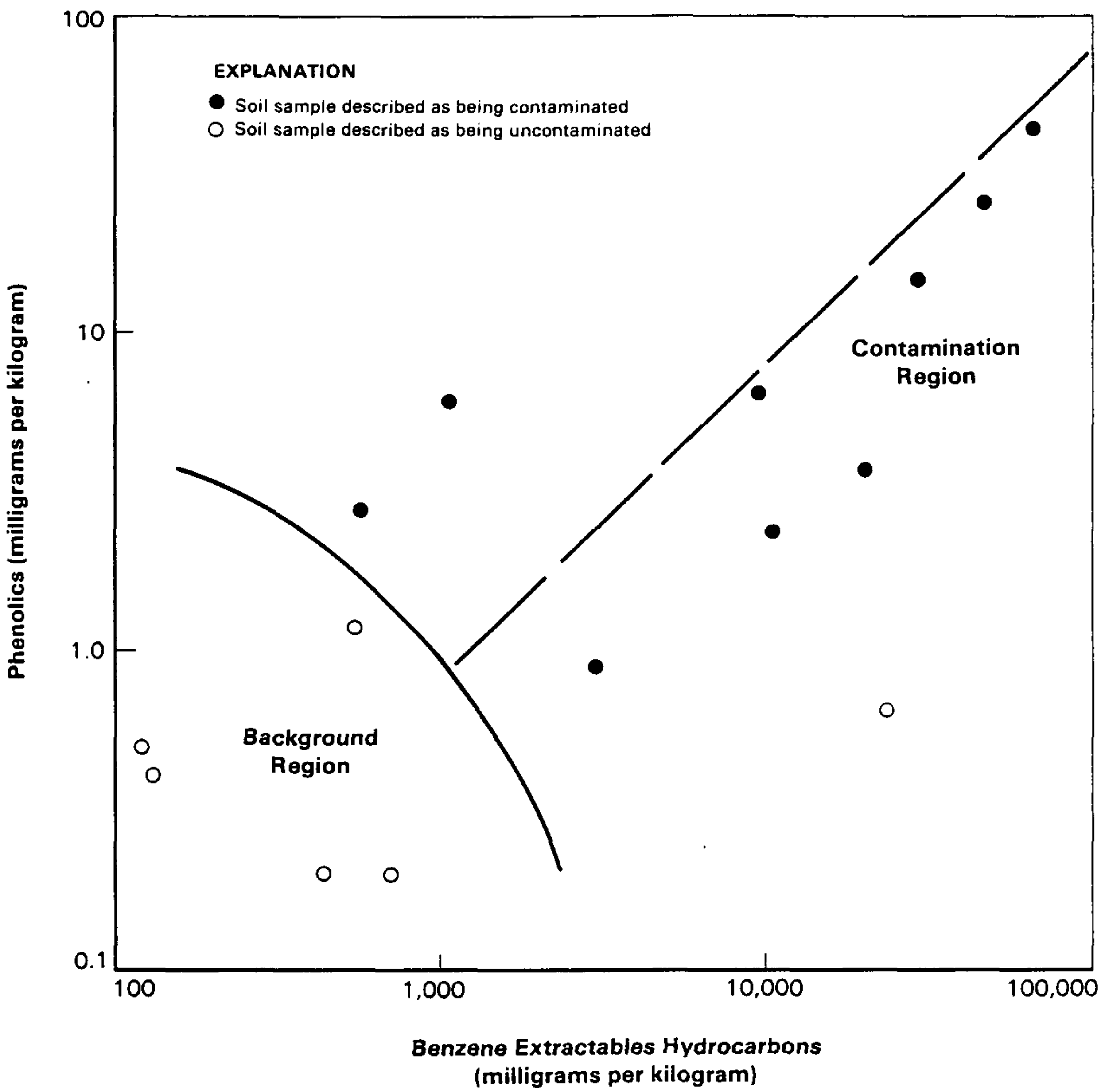
The pattern of discrete occurrences of contamination is also typical of sites at which contaminating constituents have been applied to the surface in an uneven manner. Contaminant inflow to the surficial sediments was concentrated in areas such as the coal-tar refinery, timber-treatment area, the surface-water runoff channel and the deeper surface-water flow zones of the bog. The uneven pattern of contaminant application together with the inhomogeneous and anisotropic (varying texture and response to fluid flow) properties of the sediments create uneven and difficult to predict patterns of contamination and migration of contaminants. This characteristic of uneven accumulation of contaminants is further augmented by the physical properties of the contaminants. They are variably denser than water, and they have relatively low solubilities. Their sorption/desorption properties are not well known. They are capable of migrating in patterns and at rates that are not wholly dependent on the movement of ground water, but are more dependent on their viscosity and other physical and chemical interactions with the sediments, ground water and microorganisms.

Soil contamination at the site is generally found at the original surface of the site to a depth of less than 10 feet. Below this depth, contamination is found in discrete zones that are local accumulations of contaminants which resulted from migration of these contaminants in response to their own physical properties, where and in what quantity they were applied, the nature of surface- and ground-water flow, and the physical properties and pattern of occurrence of the surficial deposits.

The occurrence and nature of surface contamination of the site has been greatly modified since RT&CC ceased manufacturing operations and the buildings were razed in 1972. Hult and Schoenberg (1981) state that "some of the most highly contaminated material above the water table was removed by shallow excavation on the site after the plant was closed in 1972". The southern half of the plant site has been converted to a park. No evidence of contamination is discernable. Dense sod covers this area; there are no signs of vegetal stress, odors or other manifestations of potential surface contamination. A 98,000 cubic yard stockpile occupies the west side of the southern half of the plant site. It is composed of spoil from the runoff collection pond and spoil from excavations at the condominiums. The quality of the stockpile soil and its affect on ground-water and subsurface soil quality are unknown. The bog, for the most part, has been filled with soil of unknown quality and much of it has been developed. As of November 1982 there are no exposures (at the surface) of contaminated soil on site except in the sump excavated next to well W23.

B5.1.5 Relationship Between Phenolic and Benzene Extractable Hydrocarbon Concentrations

Rough trends can be discerned from the graphical illustration of phenolics vs. benzene extractable hydrocarbons that appear in Figures B5-6 and B5-7. Separate plots have been constructed for artificial fill and bog deposits since the magnitude of phenolic and benzene extractable hydrocarbon concentrations vary by an order of magnitude between these types of deposits. The figures show a relative concentration of samples in the lower left corner that is qualitatively described as being not contaminated. This area is graphically representative of low phenolics and benzene extractable hydrocarbon concentrations. These samples lie within the estimated background concentration region or range of concentrations. Samples that are generally qualitatively described as contaminated fall to the right and higher than the background region. This is indicative of elevated phenolics and benzene extractable hydrocarbon concentrations. Although the points on these graphs are scattered, there is a rough trend of generally increasing phenolics concentrations with increasing benzene extractable hydrocarbon concentrations. In addition, it is evident that the benzene extractable hydrocarbon concentrations generally exceed the phenolics concentrations by up to three orders of magnitude (1,000 times). These trends are general and there are numerous exceptions.



Source: Barr (1977) Soil Exploration Co. (March, 1978)

Figure B5-6 Logarithmic Plot of Phenolic vs. Benzene Extractable Hydrocarbon Concentrations in Artificial Fill Samples

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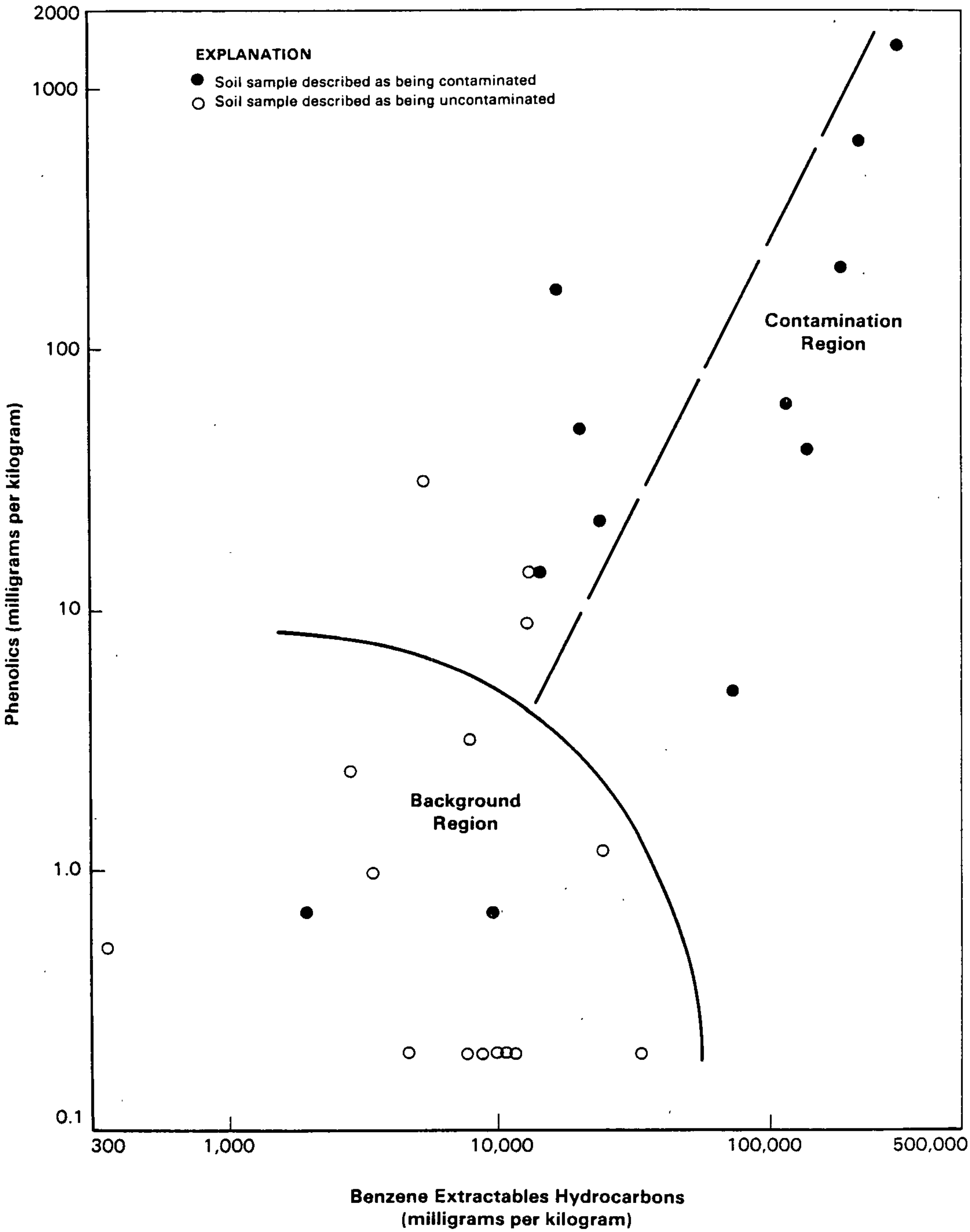

Source: Barr (1977) Soil Exploration Co. (March, 1978)

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Figure B5-7 Phenolic vs. Benzene Exractable Hydrocarbon Concentrations in Bog Deposit Samples

For example, the sampled artificial fill at boring I14 is contaminated by benzene extractable hydrocarbons (2930 milligrams per kilogram) at a level of 3.9 times the background concentration (750 milligrams per kilogram), yet the phenolic concentration (0.9 milligrams per kilogram) is approximately the same as its background concentration level (1 milligram per kilogram). At boring I3 where the phenolic concentration (2.8 milligrams per kilogram) is 2.8 times its background level (1 milligram per kilogram), the benzene extractable hydrocarbon concentration (560 milligrams per kilogram) is within its background concentration (750 milligrams per kilogram) (Figure B5-5).

These rough trends and numerous exceptions to them are the result of many factors. The source of contamination is assumed to a creosote-like material. Creosote is typically composed of about 85 per cent PAH (which is a group of benzene extractable hydrocarbons) from two to seventeen per cent phenolics and up to thirteen per cent various nitrogen- and sulphur-containing heterocyclic compounds (U.S. Forest Products Lab. 1974). The source of the phenolic and benzene extractable hydrocarbon contamination most likely contained a similar proportion of organic compounds. In addition, the benzene extractable hydrocarbon determination probably included some phenolics which would further increase the apparent relative proportion of the benzene extractable hydrocarbons. The actual composition of the source material is not known, however, and it may have varied significantly in proportions of constituents. The manner in which the surface of the site received these contaminants varied greatly among, for example, pipe leaks at the refinery, precipitation wash-off from creosoted lumber and aqueous solutions and suspensions from waste and runoff streams. These differing manners of application would directly affect the proportions of constituents. Once the contaminants had been received on the surface they were immediately subjected to dilution, transport and degradation which would further perturb the original proportion of constituents.

B5.2 Nature and Extent of Contamination in Surficial Ground Water

B5.2.1 Available Data

Erhlich et al. (1982) compiled a detailed interpretation of the nature and extent of ground-water contamination in the surficial deposits. Their sampling and analytical techniques are well documented and consistent which maximizes the reproducibility and internal consistency of the data. In addition, their interpretation is integrated with the site-area hydrogeological setting. The following indicators of contamination were used in this interpretation of the surficial ground-water contamination:

Dissolved Solids
Sodium
Chloride
Total Organic Carbon
Total Recoverable Phenol
PAH
Naphthalene

Analyses for these parameters were performed on ground-water samples from a series of wells and piezometers (narrow diameter wells) arranged in an array that runs west to east subparallel to Highway 7 (Figure B5-8). The results of these analyses are integrated with the geology and hydrology of the site area to produce a comprehensive interpretation of the nature and migration of contamination along the section defined by the linear array of wells.

Hult and Schoenberg (1981) compiled a detailed description of the hydrogeology and contamination of the Drift-Platteville aquifer in the site area. Their investigative and analytical methods are fully documented and consistent. Twenty five ground-water samples and four soil samples were analyzed. The ground-water samples were analyzed for numerous parameters including:

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[redacted]

Total Organic Carbon

Suspended Organic Carbon

Dissolved Organic Carbon

Sodium

Chloride

Nitrate

Ammonia

Phenolic Compounds, as Phenol

Selected PAH

In addition, data were compiled on the other primary and secondary drinking-water parameters. Ground-water samples were withdrawn from a broad array of wells in the site area which included two background well locations (W100 screened in the Platteville and W2 screened in the Middle Drift). The rest are on-site or downgradient wells. These wells are among those illustrated on Figures B5-9 and B5-10.

Hickok (1981) reviewed limited data on benzo (a) pyrene concentrations (a specific PAH) which were compiled by others in order to formulate an interpretation of the extent of surficial ground-water contamination.

Barr (1977) performed the initial investigation of ground-water quality in the surficial deposits. Barr's investigation included installation of 17 wells and three piezometers, and analysis of samples from these wells. These samples were analyzed for the following parameters:

Phenolics

Biochemical Oxygen Demand

Chemical Oxygen Demand

Total Dissolved Solids

Total Organic Carbon

Freon Extractable Compounds

Benzene Extractable Compounds

NON-RESPONSIVE

[redacted]

NON-RESPONSIVE

[redacted]

In addition, concentrations of several metals from the EPA primary and secondary drinking-water standards parameters were measured. Limited analyses were performed by EPA that were directed at measuring concentrations of PAH and hydroxolated aromatic hydrocarbons. Barr's sampling and analytical methods are well documented. Barr's data were used to prepare generalized contours of phenolic concentrations in the surficial deposits.

B5.2.2 Contamination Criteria for Surficial Ground Water

Wells W1, W2, W7 and W100 are considered background wells; that is they are wells that produce samples of ground water that are unaffected by sources of contamination on the site (Figure B5-9 and B5-10). Wells W1 and W100 are screened (open to the formation) in the Platteville Limestone. Wells W2 and W7 are screened in the Middle Drift. These wells lie upgradient from the site, and therefore contaminants migrating from the site would move away from these wells. Analytical data from these wells have been selected as being representative of the quality of the site-area ground water that has not been affected by contaminants migrating from the site. Downgradient wells that yield samples containing concentrations of indicator compounds greater than the concentrations in the background wells are considered contaminated.

The indicator parameters that have been selected for this interpretation of the contamination of surficial ground water are phenolics and PAH. At least one of these groups of compounds has been of major concern in most of the past studies (Barr 1977, Hickok 1981, Hult and Schoenberg 1981, Ehrlich et al. 1982). Both groups of compounds contain constituents of coal tar, although they are by no means peculiar to coal tar.

Examination of the phenolic data reported for wells W1, W2 and W7 on Table B5-1 indicated that the background concentration ranges from less than 2 to 5 micrograms per liter. This background range is probably representative of only naturally occurring levels, that is non-man-induced background levels. The land use to the northwest of the site, and therefore upgradient, is generally non-commercial. This results in minimal anthropogenic contribution to background levels.

TABLE B5-1

PHENOLIC CONCENTRATIONS[1] FROM WELLS SCREENED IN THE DRIFT AND PLATTEVILLE AQUIFERS (all concentrations in micrograms per liter)

Well No.	12/73 to 2/74(2)	4/1 to 4/12/76	9/24/76	12/9 & 12/10/76	2/18 & 4/19/77	5/26/77	5/26/77(2)	6/2/77	6/22/77	3/22 to 3/29/79	4/3 to 4/17/79	5/14 & 5/22/79	6/13 & 7/17/79	1/9 & 1/11/80	7/80	2/81
W1	-	<2	-	-	-	<2	<2	-	-	<2	-	-	-	-	-	-
W2	-	<2	-	-	-	<2	<2	-	-	5	-	-	-	-	5	-
W3	-	-	-	-	-	<2	-	-	-	-	-	-	-	-	-	-
W5	-	153	-	-	-	22	35	28	-	9	-	-	-	-	-	-
W6	-	43	-	-	-	88	190	50	-	-	93	-	-	-	-	-
W7	-	<2	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W8	-	<2	-	-	-	<2	<2	-	-	-	9	-	-	-	-	-
W9	-	3,000	-	-	760	600	1,100	600	-	110	-	-	-	-	-	-
W10	-	<2	-	-	-	4	17,000*	-	-	-	4	-	-	-	-	-
W11	-	-	-	22	-	4	23	-	-	-	3	-	-	-	-	-
W12	-	-	-	14	-	-	-	-	-	-	26	-	-	-	400	-
W13	-	-	-	-	4,800	51,000	56,000	49,000	50,000	81,000	-	-	-	-	29,000	21,000
	-								27,000							
W15	-	-	-	-	-	28	37	-	-	-	-	-	-	-	-	-
W16	-	-	-	-	2	4	<2	-	-	-	<2	-	-	-	-	-
W17	-	-	-	-	280	140	340	180	32	-	240	-	-	-	-	-
W18	-	-	-	-	-	-	-	-	-	73	-	-	-	-	-	-
W19	-	-	-	-	-	-	-	-	-	10	-	-	-	-	-	-
W20	-	-	-	-	-	-	-	-	-	34	-	-	-	-	-	-
W22	-	-	-	-	-	-	-	-	-	<2	-	-	-	-	-	-
W26	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-
W27	-	-	-	-	-	-	-	-	-	-	-	-	52	-	-	-
W30	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W33	1,000-1,400	170	-	-	-	140	390	-	-	-	-	-	220	-	-	-
W38	-	-	-	-	-	-	-	-	-	-	-	-	-	11.6	-	-
W41	-	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-

TABLE B5-1 (Continued)

Well No.	12/73 to 2/74(2)	4/1 to 4/12/76	9/24/76	12/9 & 12/10/76	2/18 & 4/19/77	5/26/77	5/26/77(2)	6/2/77	6/22/77	3/22 to 3/29/79	4/3 to 4/17/79	5/14 & 5/22/79	6/13 & 7/17/79	1/9 & 1/11/80	7/80	2/81
W60	-	-	-	-	-	-	-	-	-	-	-	5.8	4.8	-	-	-
W75	-	-	-	-	-	-	-	-	-	-	-	<2	-	-	-	-
W100	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
W101	-	-	-	-	-	-	-	-	-	-	14	-	-	-	-	-
W107	-	-	-	-	-	-	-	-	-	-	-	-	-	11.6	-	-
W115	-	-	-	-	-	-	-	-	-	-	9	-	-	-	-	-
W116	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-
W117	-	-	-	-	-	-	-	-	-	-	20	-	-	-	10	<40
P14	-	-	-	-	-	-	-	-	-	-	-	-	-	-	8,000	10,700
P119	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	<90
Source:	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Barr 1977	Hult 1981	Hult 1981	MDH 1980	MDH 1980	MDH 1980	Ehrlich 1982	Ehrlich 1982

*Suspect data believed to be the result of reporting, analytical, or sampling error. Excluded from discussions on phenolic contamination of Drift-Platteville ground water.

(1)Data reported as phenolics or phenols unless otherwise noted. Data originally reported as milligrams per liter.
(2)Data from Minnesota Department of Health (1977) as reported by Barr.
- Indicates no analysis of phenolics conducted.

Land use at and to the southeast of the site is generally commercial or mixed residential and commercial. For purposes of characterizing the level of contamination emanating from the site, it is more appropriate to define a background concentration that represents a combination of natural and anthropogenic sources.

Data on phenolic concentrations from wells W8, W10, W16, W19 and W20 which are screened in the Drift and Platteville were examined for evidence of anthropogenic contribution to the background concentration that is representative of a commercial area. These wells lie to the south of the site and are laterally removed across the hydraulic gradient from the site. The mapped easterly directed hydraulic gradient indicates that ground-water borne contaminants could not flow from the site to these wells (Hult and Schoenberg 1981). Phenolic concentrations from these wells range from less than 2 to 34 micrograms per liter. Based on these limited data, a total background concentration on the order of 10 micrograms per liter is taken as representative of the commercial portions of the site area.

Background concentrations of PAH have been derived from a limited set of analyses performed on samples from wells W1, W2 and W100 (Table 5-2). An analyses performed on a sample from well W1 (screened in the Platteville to the north of the site) produced a combined fluoranthene and pyrene (specific PAH) concentration of 0.45 micrograms per liter. Analyses performed on samples from well W100 also screened in the Platteville to the north of the plant site produced a total PAH concentration range of 0.005 to 0.063 micrograms per liter. A set of analyses performed on samples from well W2 (screened in the Middle Drift) produced total PAH ranging from 0.02 to 0.69 micrograms per liter (CH2M Hill 1982). Anthropogenic components to the background that are evident in wells W10, W16, W19 and W20 elevate the site-area PAH background concentration to a range of 0.018 to 3.1 micrograms per liter. Based on these data a maximum background concentration of 3 micrograms per liter was chosen. Unfortunately the detection limit for total PAH reported by Ehrlich et al. (1982) is 40 micrograms per liter for analyses on samples from downgradient wells.

TABLE B5-2

PAH CONCENTRATIONS FROM WELLS SCREENED IN THE DRIFT AND PLATTEVILLE AQUIFERS

USGS Well No.	Sample Date	Acenaphthene	Acenaphthylene	Anthracene	Benzo (a) Anthracene	Benzo (j) Fluoranthene	Benzo (k) Fluoranthene	Benzo (g,h,i) Perylene	Benzo (a) Pyrene	Benzo (e) Pyrene	Chrysene	Dibenzo (a,c) Anthracene	Dibenzo (a,h) Anthracene	Fluoranthene	Fluorene	1-Methyl-Naphthalene	2-Methyl-Naphthalene	Naphthalene	Perylene	Phenanthrene	Pyrene	1,2,3,4-Tetra-Hydropyrene	9,10-Benzphenanthrene	O-Phenylenepyrene	Total Measured PAH	Reference
W1	2/6/81	<0.3	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	0.24	<0.35	-	-	-	-	<0.25	0.21	-	-	-	0.45	MRI 1981
W2	6/30/80	-	-	-	-	0.012	-	-	-	-	-	-	-	0.0048	-	-	-	-	-	0.0032	-	-	-	-	0.02	CH2M 1982
W2	2/6/81	<0.3	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	0.41	<0.35	-	-	-	-	<0.25	0.28	-	-	-	0.69	MRI 1981
W2	9/8/82	<0.0013	<0.001	<0.001	<0.001	-	<0.001	<0.001	<0.001	-	<0.001	-	<0.0014	0.0035	0.0014	0.0012	0.002	0.0082	<0.001	0.011	0.0046	-	-	<0.0017	0.032	CH2M 1982
W2	11/2/82	<0.0013	<0.001	<0.001	<0.001	-	<0.001	<0.001	<0.001	-	<0.001	-	<0.0014	0.001	<0.0014	0.0012	0.0052	0.0086	<0.001	0.001	0.001	-	-	<0.0017	0.018	CH2M 1982
W3	-																									
W5	-																									
W6	-/79	-	1,000	400	400	-	-	-	200	-	400	-	-	2,000	2,000	-	1,000	-	-	5,000	<8,000	-	-	-	12,400	Hult 1981
W7	-																									
W8	-																									
W9	-																									
W10	-/79	-	<2.0	<0.09	<0.5	-	-	-	<0.1	-	<0.6	-	-	0.5	2.0	-	<2.0	-	-	0.6	<1.0	-	-	-	3.1	Hult 1981
W11	-/79	-	<4.0	0.2	<3.0	-	-	-	0.1	-	<0.4	-	-	0.8	1.0	-	<2.0	-	-	2.0	<0.8	-	-	-	4.1	Hult 1981
W12	-																									
W13	-/80*	-	72,000	68,000	-	-	-	-	-	-	-	-	-	84,000	48,000	32,000	56,000	310,000	-	156,000	60,000	-	-	-	970,000	Ehrlich 1982
W13	4/27/81	100,000	<0.25	-	-	-	<0.8	<1.3	<0.8	-	200,000	-	-	600,000	200,000	-	-	1,400,000	-	800,000	300,000	-	-	-	3,600,000	MRI 1981
W13	2/81**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	15,500	-	-	-	-	-	-	22,300	Ehrlich 1982
W13	6/81	+	<5,250	66,000	300,000	-	-	92,000	160,000	-	<360	-	16,000	420,000	110,000	-	-	220,000	-	630,000	500,000	-	-	27,000	2,541,000	USGS 1981
W13	6/81	8,000	4,000	2,000	4,000	-	<2,000	<2,000	-	-	4,000	-	-	4,000	40,000	4,000	120,000	230,000	2,000	110,000	2,000	-	2,000	-	531,000	USGS 1981
W13	9/9/82	200,000	-	57,000	59,000	-	38,000	-	-	-	44,000	-	-	240,000	140,000	79,000	210,000	980,000	-	560,000	160,000	-	-	-	2,787,000	CH2M 1982
W15	-																									
W16	-/79	-	<2.0	<0.09	<0.4	-	-	-	<0.1	-	<0.6	-	-	<0.04	0.1	-	<2.0	-	-	<0.2	<1.0	-	-	-	0.1	Hult 1981
W17	-/79	-	<2.0	0.2	<0.4	-	-	-	<0.1	-	<0.6	-	-	0.2	4.0	-	<2.0	-	-	0.6	<1.0	-	-	-	5.0	Hult 1981
W18	-																									
W19	-/79	-	0.008	<0.0006	<0.002	-	-	-	<0.004	-	0.006	-	-	0.0005	<0.006	-	<0.01	-	-	0.004	<0.008	-	-	-	0.018	Hult 1981
W20	-/79	-	0.02	0.002	<0.003	-	-	-	<0.004	-	0.004	-	-	0.0008	0.01	-	<0.01	-	-	0.004	<0.008	-	-	-	0.048	Hult 1981
W22	-																									
W27	7/17/79	-	-	-	<0.0053	-	-	-	<0.008	-	<0.006	<0.0048	-	0.095	-	-	-	-	<0.0022	<0.0034	7.9	-	-	<0.0032	8.0	MDH 1980
W30	4/26/79	-	0.035	<0.01	<0.002	-	-	-	<0.002	<0.005	<0.005	<0.002	-	0.0017	-	-	<0.01	-	<0.001	0.0073	0.026	<0.02	-	0.001	0.07	MDH 1980
W30	5/14/79	-	0.069	<0.01	0.003	-	-	-	0.0029	<0.005	<0.005	<0.002	-	0.0036	-	-	<0.01	-	<0.001	0.0098	0.028	<0.02	-	0.0017	0.093	MDH 1980
W33	6/5/79	-	<0.01	<0.01	0.0026	-	-	-	<0.002	<0.005	<0.005	<0.002	-	0.0041	-	-	<0.01	-	<0.001	<0.005	<0.01	<0.02	-	0.004	0.011	MDH 1980
W38	1/9/80	-	6.3	<0.07	2.0	<0.5	<0.14	<0.75	<0.45	0.31	8.2	<0.44	-	0.65	<0.43	-	16.0	-	<0.1	3.1	<0.86	6.8	<0.11	0.45	43.81	MDH 1980
W38	2/24/81	<0.3	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	2.1	<0.35	-	-	-	-	<0.25	1.4	-	-	-	3.50	MRI 1981
W41	-																									
W60	6/13/79	-	<0.01	<0.01	0.0096	-	-	-	0.0093	0.012	<0.005	<0.002	-	0.0085	-	-	<0.01	-	<0.001	<0.005	<0.01	<0.02	-	0.0046	0.044	MDH 1980
W75	5/22/79	-	0.015	<0.01	<0.002	-	-	-	<0.002	<0.005	<0.005	<0.002	-	0.0028	-	-	<0.01	-	<0.001	<0.005	<0.01	<0.02	-	0.0024	0.020	MDH 1980
W100	-/79	-	<0.003	0.0008	0.001	-	-	-	-	-	<0.0006	-	-	0.001	0.04	-	0.01	-	-	0.01	<0.008	-	-	-	0.063	Hult 1981
W100	6/30/80	-	-	-	-	-	-	-	-	-	-	-	-	0.0018	-	-	-	-	-	0.0049	-	-	-	-	0.007	CH2M 1982
W100	7/15/80	-	-	-	-	0.0019	-	-	-	-	-	-	-	0.001	-	-	-	-	-	-	-	-	-	-	0.005	CH2M 1982
W100	9/9/82	<0.0065	<0.005	<0.005	<0.005	-	<0.005	<0.005	<0.005	-	<0.005	-	<0.007	<0.005	<0.007	<0.006	<0.01	<0.0095	<0.005	<0.005	<0.005	-	-	0.0085	-	CH2M 1982
W100	11/2/82	<0.0013	<0.001	<0.001	0.001	-	<0.001	<0.001	<0.001	-	0.0015	-	<0.0014	0.0027	<0.0014	0.0012	0.005	0.0093	<0.001	0.001	<0.001	-	-	0.0017	0.022	CH2M 1982
W101	-/79	-	0.4	<0.001	0.001	-	-	-	<0.004	-	<0.0008	-	-	0.001	0.04	-	0.6	-	-	<0.004	0.01	-	-	-	1.042	Hult 1981
W101	2/6/81	0.84	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	<0.25	<0.35	-	-	4.7	-	<0.25	0.3	-	-	-	5.54	MRI 1981
W107	-																									
W115	-/79	-	<0.002	<0.0001	<0.0006	-	-	-	<0.004	-	<0.0008	-	-	0.001	0.02	-	0.02	-	-	<0.0002	0.02	-	-	-	0.061	Hult 1981
W116	2/6/81	<0.3	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	<0.25	<0.35	-	-	-	-	<0.25	<0.3	-	-	-		MRI 1981
W117	2/81	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	<40	-	-	-	-	-	-		Ehrlich 1982
W117	2/6/81	2.5	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	<0.25	<0.35	-	-	-	-	<0.25	<0.3	-	-	-	2.5	MRI 1981
W117	3/11/81	3.0	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	<0.25	<0.35	-	-	-	-	<0.25	<0.3	-	-	-	3.0	MRI 1982
W117	6/81	4.9	<0.02	<0.001	<0.003	-	-	<0.007	<0.004	-	<0.001	-	<0.012	<0.007	<0.002	-	-	<0.018	-	<0.001	0.006	-	-	0.002	4.9	USGS 1981
W117	6/81	2.0	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	<1	-	-	-	-	-	-	2	USGS 1981
M14	3/11/81	160	<0.25	-	-	-	<0.8	<1.3	<0.8	-	<0.5	<1.4	-	<0.25	80	-	-	50	-	30	<0.3	-	-	-	320	MRI 1981
M14	2/81	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11,200	-	-	-	-	-	-	11,200	Ehrlich 1982
M14	6/81	20	<2.0	0.10	<0.12	-	-	<0.29	<0.18	-	<0.14	-	<0.27	<0.3	6.3	-	-	270	-	1.6	0.85	-	-	0.09	299.05	USGS 1981
M14	6/81	20	-	-	-	-	-	-	-	-	-	-	-	-	40	-	400	850	-	100	-	-	-	-	1,410	USGS 1981
M19	2/81	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2,300	-	-	-	-	-	-	2,400	Ehrlich 1982

< Less than detectable limit.
- Not analyzed for or not reported.
+ Measurement interference.
* Data specifically reported to be from fluid phase.
** Data specifically reported to be from aqueous phase.

B5.2.3 Review of Surficial Ground-Water Contamination Interpretations

Barr (1977), Hult and Schoenberg (1981) and Ehrlich et al. (1982) have prepared comprehensive interpretations of surficial ground-water contamination. These are summarily discussed, and they have been integrated into this interpretation.

Barr (1977) compiled two interpretations of surficial ground-water contamination based on phenolic concentrations in the wells that they installed and in two private wells (Midco Well W33 and Hartmann Well W41) that are located east of the site (Figure B5-10). These two private wells were screened in the Platteville, Glenwood and St. Peter bedrock units. That is, the well bore was cased through the surficial deposits, but open to these underlying bedrock units.

Barr's (1977) phenolic contours outline a northwest-southeast trending elliptical area of surficial ground-water contamination that is centered on W13. It extends half way up the plant site to the northwest and approximately 2,000 feet to the east of W13. The contouring is misleading in that it portrays a smooth and steady decrease of phenolic concentrations areally away from W13 and constant concentration with depth. This is not the case. The data indicate widely varying concentrations with depth and distance. For example, at the nested wells W11 (screened in the Middle Drift) and W17 (screened in the Lower Drift) a series of three analyses indicate that the phenolics concentrations in the Middle Drift (4 to 23 micrograms per liter) are consistently a full order of magnitude less than the concentration in the Lower Drift (140 to 340 micrograms per liter). The variation in concentration is the result of the complex geology which restricts movement of ground water and contaminants, the general flow of ground water which is directed to the east with a downward component, and the physical and chemical properties of the contaminants.

Barr's general interpretation is consistent with the data and is substantiated by Hult and Schoenberg's (1981) and Ehrlich's et al. (1982) more detailed investigations. Barr's data indicate that the surficial ground water is contaminated by phenolics starting under the southern half of the plant site, continuing under the bog (rising to the greatest levels of contamination here) and dissipating away from the bog in an easterly direction. Also the pattern of contamination descends in an easterly direction. The areal limits of contamination are clearly defined, except to the east, by wells in which phenolics were not found above the detection limit (2 micrograms per liter).

The Midco well (W33) lies approximately 1,500 feet east of the center of the bog. Sampling indicated that it was contaminated in concentrations ranging from 140 to 390 micrograms per liter of phenolics. This well was screened in the Platteville, Glenwood and St. Peter bedrock units until June, 1979. Thereafter, casing was inserted into the well to close off the Platteville and Glenwood bedrock units leaving the well open to only the St. Peter Sandstone. Prior to June, 1979 it was capable of allowing flow from the surficial deposits and Platteville Limestone into the St. Peter Sandstone. Well W17 (screened near the lower drift and Platteville contact) is also contaminated by phenolics ranging in concentration from 32 to 340 micrograms per liter. Wells further upgradient toward the bog are increasingly contaminated with phenolics. This pattern depicts a former link between the high level of phenolics found in well W13 below the bog and local phenolic contamination in well W33 which was screened in the Platteville, Glenwood, and St. Peter bedrock units when these analyses were performed.

Hult and Schoenberg (1981) predominantly rely on inorganic and organic indicator parameters such as sodium and dissolved organic carbon to define a plume of contamination emanating from the site and continuing downgradient to the tributary bedrock valley. Their data on phenolic and PAH migration indicate that these substantially less soluble compounds are migrating in the same pattern as the relatively soluble constituents, but the rate and distance of migration are far less. In addition, the physical, chemical and biological processes that characterize the hydrocarbon enriched ground fluid and the aquifer medium are hypothesized to play major roles in the transport process. Investigation of these processes was beyond the scope of their paper.

The interpretation of Ehrlich et al. (1982) refines Barr's (1977) and Hult and Schoenberg's (1981) contaminant migration descriptions and details the mechanisms of attenuation of phenolics. Ehrlich et al. (1982) profile phenolic concentrations from well W13 to a point approximately 7,500 feet to the east. In a manner similar to Barr (1977), Ehrlich et al. (1982) describe an eastward trending and descending pattern of decreasing concentrations of phenolics. Phenolic concentrations drop to below the detection limit (less than 90 micrograms per liter) in the Middle Drift at a point 1,400 feet to the east of well W13. Since contaminants descend with distance from the site, contamination would not be detectable in the Middle Drift, but it would be in the lower drift or Platteville. Below the Platteville, the Glenwood Shale would impede further downward migration. This was apparently the case at the Midco well (W33) which is 1,500 feet downgradient from the site; it took in water from the Platteville rather than the overlying surficial units which at this distance downgradient from the site are no longer contaminated.

PAH concentrations dropped to below the detection limit (40 micrograms per liter) in the Middle Drift 4,500 feet to the east of the site, past the tributary bedrock valley in well W117 (Ehrlich et al. 1982). Over this distance from the bog, however, the contribution of PAH from other sources has not been addressed, and based on examination of the type and density of commercial and industrial activity there are many potential contributors of PAH.

Ehrlich's et al. (1982) investigation reveals that phenolics are in part anaerobically biodegradable, (that is, natural soil microorganisms convert phenolics to methane and carbon dioxide in an oxygen defficient environment such as ground water) at least in the Middle Drift, to background levels within 1,400 feet downgradient from the site. The full degradation and attenuation process is not yet understood. Naphthalene (the predominant PAH constituent of the aqueous phase in well W13) attentuates at a rate of approximately 20,000 micrograms per liter to 2,000 micrograms per liter over 1,400 feet and 20,000 micrograms per liter to below the detection limit (40 micrograms per liter) over 4,500 feet downgradient. The mechanisms of naphthalene attenuation are not yet understood.

These attenuation studies reveal that over a 65 year period phenolics have migrated less than 1,400 feet downgradient through the surficial deposits from the site, and naphthalene has potentially (because other sources have not been addressed) migrated less than 7,500 feet.

B5.2.4 Nature and Extent of Two-Phase Ground Fluid at Well W13

The peculiar nature of the ground fluid in well W13 has been addressed by Barr (1977) Hult and Schoenberg (1981) and Ehrlich et al. (1982), and in each case this ground fluid has been described as a potential source of downgradient ground-water contamination. Well W13 is screened in a unique zone of ground fluid. The pore fluid occupying the interstices of the Middle Drift deposits around the well screen is a two phase fluid. One phase is an aqueous solution; the other is an oily, PAH enriched phase. The aqueous phase contains 29,100 micrograms per liter phenolics and 22,300 micrograms per liter PAH (Ehrlich et al. 1982). The mixed phases contain nearly one gram of PAH per kilogram of fluid, or nearly 0.1 per cent. Detailed gas-chromatography analyses revealed that the PAH contained in the fluid are present in types and proportions that are characteristic of creosote (Ehrlich et al. 1982).

The volume of the Middle Drift that contains a two-phase pore-fluid can be estimated from the boring logs and chemical analyses. Boring I9 which was drilled at the location of well W13 produced a sample of sand and gravel at approximately 48 feet that was described as visibly contaminated with black oily fluid. This is the same depth at which Well W13 is screened. The benzene extractable hydrocarbon concentration was 1,740 milligrams per kilogram (1.7 times background); and the phenolic concentration was 2.3 milligrams per kilogram (2.3 times background) as shown in Figures B5-2 and B5-4. Samples of the Middle Drift taken above these visibly and analytically contaminated samples which were taken from depths of 30 feet to 41 feet were not described as visibly contaminated, and the analyses produced benzene extractable hydrocarbon and phenolic concentrations that were well within background concentrations. The gray clay layer at a depth of 49 feet beneath the Middle Drift is also contaminated by phenolics which are present in a concentration of 7.8 milligrams per kilogram (7.8 times background). Benzene extractable hydrocarbons were present in a concentration of 170 milligrams per kilogram which is well within background limits. The remaining five samples taken to the bottom of boring I9 were not visibly or analytically contaminated.

The vertical extent of the two-phase pore-fluid correlates with the visibly and analytically contaminated soil in the Middle Drift in boring I9. Based on this correlation the two-phase pore-fluid occupies an approximately five foot thick lens on top of the upper till. In nearby boring G13 (70 feet to the east) distinctly elevated concentrations of oil, grease and paraffin (27,000 milligrams per kilogram or 2.7 per cent) and phenolics (15.5 milligrams per kilogram) were detected in the Middle Drift at a depth of 46 feet (Soil Exploration Co. 1974). It is reasonable to correlate these data with those from I9. Borings G13, E5, G8 and I11 further to the northeast do not show patterns of contamination that would correlate to that in borings I9, G13 and well W13. Boring I11 does show a small zone of possible light phenolic contamination (1.1 milligrams per kilogram or 1.1 times background) just above a clayey sand lens at a depth of 30 feet. To the southwest boring I10 contains no indication of visible contamination. There is, however, a small zone of phenolic contamination associated with the upper till at a depth of 40 feet. The phenolic concentrations are 2.4 and 3.1 milligrams per kilogram or 2.4 and 3.1 times the background concentration. To the northwest in boring I12 the soil is continuously contaminated from a depth of four feet through a depth of 20 feet or eight feet below the glacio-lacustrine clay into the glacio-fluvial deposits (Middle Drift). Below this depth, both benzene extractable hydrocarbon and phenolic concentrations are within background levels. In addition, no visible contamination is reported. None of the wells closest to W13 (W5, W8, W9) produce two-phase ground-fluid.

The pattern of soil and ground-water contamination around W13 indicates that the two-phase ground-fluid occurs in an approximately five feet thick lens that extends approximately 100 feet in all directions from W13. These estimated dimensions probably represent a maximum volume, and the two-phase ground-fluid may occupy a far smaller volume. Assuming that this ground fluid occupies a right circular cone five feet high and 100 feet in radius and the Middle Drift sediments in which the two-phase ground fluid occurs has a porosity of 30 per cent, this ground fluid occupies a maximum volume of 16,000 cubic feet. The thickness of the lens and the level of contamination decline toward the edges of the lens. The light contamination shown in boring I10 as a zone at 40 feet may represent the distal expression of the lens of contamination associated with the two-phase pore-fluid encountered in well W13. In addition, the interaction between the ground-water flux around and probably through this zone is not known.

This zone of two-phase ground fluid probably formed as a result of the greater density of the contaminants relative to the ground water. The contaminants were able to descend through the water-saturated bog sediments. The mechanism of agglomerating the contaminants into this particular volume of the Middle Drift is not known. That this zone lies below the fine-grained glacio-lacustrine deposit indicates that either this deposit is peculiarly permeable to contaminants or more likely that it is locally discontinuous near well W13. This well is within 200 feet of the Highway 7 embankment. It is not known whether excavation preceeded placement of the embankment fill. If excavation of the bog deposits was carried out, it is likely that the thin glacio-lacustrine deposits were often breached along the alignment of the fill. These breaches would provide a conduit of downward contaminant migration from constituents contained in the bog down to the Middle Drift sediments.

It is also possible that the drilling of well W13 and boring I9 introduced or accelerated the downward movement of coal-tar constituents. Examination of the 1945 aerial photographs indicates that well W13 and boring I9 were drilled in a deeper area of the bog near the end of the Highway 7 culvert discharge to the south side of the bog. It is likely that the heavier contaminants accumulated in this deeper portion of the bog. Puncturing of this contaminant accumulation and the underlying sediments, including the glacio-fluvial sediments, could have provided a means of at least accelerating the downward migration of the contaminant into the Middle Drift sediment.

B5.2.5 Constituent Migration into the St. Peter Aquifer

Analytical studies of the surficial ground water indicate that the site acts as a local source of contamination to the Drift-Platteville aquifer, and that this contamination is moving off site. Sampling and analysis indicate that the Midco well (W33) (which lies 1,500 feet downgradient) was a potential conduit of contaminant migration from the surficial deposits and Platteville Limestone into the St. Peter Sandstone. Whether contaminant migration took place via the Midco well (W33) cannot be determined because there is too little information on the pre-1979 operation of the well, hydraulic parameters in the sediments and rock formations at the well, and the length of time that contaminants have been within the zone of influence of the well. In addition, Hult and Schoenberg (1981) report that a down-hole flow-meter measurement in well W33 did not detect any downward flow from the Drift-Platteville to the St. Peter. The Hartmann well (W41) lies 2,500 feet downgradient from the site. It is constructed such that flow from the Platteville down into the St. Peter could potentially occur. Phenolic concentrations at that well, however, are reported to be 2 micrograms per liter which is within background concentrations. The Terry Excavating Well (W27) is located several hundred feet to the east of the southern half of the plant site. It was, until October, 1979, screened in the Platteville, Glenwood and St. Peter bedrock units. It is documented that well W27 was at that time grouted up to the Platteville thereby sealing off the St. Peter (Minnesota Department of Health 1981). Prior to reconstructing well W27, the Minnesota Department of Health (1979) measured a phenolic concentration of 52 micrograms per liter in this well which is five times above the background concentration. Although the tributary bedrock valley which lies 3,000 feet downgradient from the site has been documented to be an area of ground-water inflow to the St. Peter from the surficial deposits and Platteville Limestone, no phenolic or PAH analytical data are available from the valley itself or from the Lower Drift and Platteville between the Midco well (W33) and the tributary bedrock valley. The available data from Barr (1976 and 1977), Hult and Schoenberg (1981) and Erhlich et al. (1982) indicate that phenolics are not entering the tributary bedrock valley, but are being degraded and otherwise attenuated to background levels before reaching the valley.

The PAH data from Ehrlich et al. (1982) indicate that there is an order of magnitude (ten times) decrease in concentration (from 22,300 to 2,400 micrograms per liter) 1,400 feet downgradient from well W13 to piezometer P119 which is screened in the Middle Drift. No more data are reported between piezometer P119 and the tributary bedrock valley which lies 1,600 feet further downgradient. On the other side of the valley, PAH concentrations were reported at a range of 2 to 4.9 micrograms per liter in well W117 (7,500 feet downgradient from well W13). It cannot be determined whether attenuation or flow down into the tributary bedrock valley is responsible for the low levels of PAH at well W117. It is also probable that well W117 is screened too high in the drift to intercept any potential contaminant flow originating at the site that would, at this distance from the site, be following flow paths in the lower-most surficial deposits and Platteville. The PAH concentrations measured in well W117 are most likely the result of urban and commercial/industrial activity from nearby upgradient sources rather than on-site sources.

The data indicate that phenolics or PAH which originate at the site have not been detected in the St. Peter Sandstone. Prior to 1979 contaminated ground water in the Drift-Platteville aquifer was induced to flow into the well bores of wells W27 and W33. This contaminated ground water probably came into contact with the St. Peter Sandstone that was exposed in the open well bore. Neither hydraulic flow nor contaminant migration into the St. Peter via these shallow multi-aquifer wells has been measured. The Hartman Well (W41) is described as being open to the Platteville and St. Peter (Hult and Schoenberg 1981), however, its present status is unknown. One phenolic concentration of 2 micrograms per liter was reported from this well by Barr (1976). This concentration is within background concentration levels. Although well W41 is a potential conduit of contaminant migration to the St. Peter by virtue of its construction and downgradient location, the data indicate that phenolic contamination from the site is degrading and otherwise attenuating before it reaches well W41.

The known, shallow multi-aquifer wells that are downgradient from the site are unlikely conduits of contamination to the St. Peter, because they have been reconstructed to eliminate potential flow into the St. Peter or they lie outside the documented extent of ground-water contamination. In addition, the data do not indicate that contaminants originating at the site are migrating into the St. Peter via the tributary bedrock valleys. The potential for contaminant flow from the site to underlying bedrock aquifers via deep on-site multi-aquifer wells (W23 and W105) is discussed in Appendix D.

B6. CONCLUSIONS

This section presents the conclusions derived from interpretation of the cited data. The conclusions are directed at satisfying the objectives discussed in subsection B1.1.

B6.1 General Conclusions

The interpretation of the hydrogeology and the nature and extent of contamination at and derived from the site supports these basic conclusions:

- The upper five to ten feet of the original (pre-park construction and/or filling) surface of the southern half of the plant site and the bog are generally contaminated (Figure B6-1). The boring data indicate that below this depth, soil contamination, where it occurs, occupies discrete zones that are generally several feet thick and on the order of 100 feet in lateral extent. These zones are generally associated with lenses of fine-grained sediment which restrict the downward movement of the contaminants.
- The ground water beneath and to 1,400 feet down the hydraulic gradient (to the east) from well W13 is variably contaminated by phenolics and PAH. Figure B6-1 depicts an inferred zone of contamination that is contiguous with the site and primarily attributable to manufacturing and wood-treating processes at the plant site.

B6.2 Discussion of Conclusions

The surficial sediments underlying the southern half of the plant site received contaminants from among other sources: processes, drippings, spills and piping failures. These contaminants in part percolated down through the unsaturated sediments to the ground-water table. These contaminants, although of very low solubility, then entered the ground-water system. In this system, the dissolved contaminants were transported in an easterly and downward

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flow paths. These contaminants are denser than water, and if not dissolved would tend to descend through the ground-water saturated sediments. The Glenwood Shale which underlies the Drift-Platteville aquifer confines the downward migration of dissolved and free contaminants to predominantly lateral flow paths in a generally eastward direction.

The bog received contaminants from process wastewater and runoff from the plant site which were channeled under Walker Street into the bog. The bog has been generally saturated with ground water throughout the production history. Contaminants entering the bog were immediately incorporated into the water system of the bog. Some dissolved contaminants probably migrated via surface-water discharge to the south or percolated with ground water to underlying sediments. Some of the contaminants, however, were ponded in the bog above their solubilities. These contaminants could then migrate downward in response to their greater density relative to water, and influenced by other physical and chemical interactions among the contaminants, ground water and sediments. Dissolved contaminants could migrate easterly and downward with the dominant ground-water flow paths. Relatively concentrated undissolved contaminants have agglomerated into what is estimated to be at maximum an approximately five foot thick by 200 foot diameter lens in the Middle Drift at well W13 creating a two component ground fluid composed of a phenolic-rich aqueous phase and a denser PAH-rich oily phase. The mixed phases contain less than 0.1 per cent PAH (Ehrlich et al. 1982). The concentrations of contaminants also decrease toward the edges of the lens.

The bog (including the ground-fluid lens) and the southern half of the plant site were and continue to be a source of surficial ground-water contamination which includes ground water in the unconsolidated surficial deposits and the Platteville Limestone. Fresh contaminants are also introduced into the ground water via the flux of percolating precipitation and surface runoff which comes through the unsaturated contaminated sediments. In addition, the saturated contaminated sediments are to varying degrees capable of desorbing contaminants. Areas of heavy contamination will tend to keep transmitting ground water that contain dissolved contaminants.

The easterly and downward ground-water flow pattern continually supplies uncontaminated ground water from upgradient. This ground water is capable of picking up contaminants from the contaminated sediments or agglomerations of the contaminants in the saturated thickness of unconsolidated sediments which underlie the site.

Contaminants which have been picked up in this way migrate off site in an easterly and downward direction with the site area ground-water flow pattern. Contaminant migration is complicated by geologic restrictions to fluid flow, physical, chemical and biological reactions among the constituents, ground water, sediments and microorganisms. These complications slow contaminant migration to rates far less than ground-water flow.

The extent of contamination of the surficial ground water is well marked in all directions except to the northeast where there are no wells or piezometers north of Highway 7 and downgradient to the east from the southern half of the plant site except for well W27 (Terry Excavating). To the west and north wells W1, W2, W3 and W7 provide definitive background information and limits of contaminant migration in these directions. Data from wells W8, W10 and W16 indicate that the extent of site induced ground-water contamination lies to the northeast of these wells. To the east, ground-water contamination cannot be definitively traced in the Middle Drift beyond piezometer P119 or 1,400 feet to the east of the site. In the Platteville, contamination was detected at a point 1,500 feet to the east. At that point (Midco well W33), it is known that the the Platteville was contaminated prior to 1979. Migration into the St. Peter Sandstone via shallow multi-aquifer wells or the tributary bedrock valleys has not been detected.

B6.3 Future Contaminant Migration

The ground-water quality data reviewed for this interpretation do not depict any trend of change in ground-water quality that would allow prediction of a pattern of change in the ground-water quality in the near future. Phenolics, however, are known to degrade at the site in ground water by the action of anaerobic bacteria (Ehrlich et al. 1982). The pattern of phenolic degradation indicates that the ground water in the unconsolidated surficial sediments is being cleansed of phenolics within 1,400 feet of the site.

It is reasonable to assume that the rate of off-site migration of contaminants will not increase. These contaminants have been available for migration for 65 years since the start of coal-tar refining and creosoting operations and attendent process wastewater discharge. Prior to this, effluent from sugar-beet refining processes was discharged to the bog and may constitute a contribution to the source of organic constituents. The quality of wastewater discharged to the bog has improved over the last 40 years as wastewater treatment was steadily improved. For example, an API separator was installed in 1941. Since institution of wastewater treatment, the rate of constituent input to the bog has probably steadily declined to zero in 1971 when the plant closed. The availability of contaminants is decreasing since there is no longer any input and physical, chemical and biological mechanisms are steadily attenuating the contaminants in the sediments and ground water. The flux of contaminants vertically through the bog deposits into the Drift-Platteville aquifer was probably greatest during plant operation when the steady influx of wastewater exceeded both surface flow out of the bog and evapotranspiration. This created a ground-water mound and a resultant increased vertical gradient. Since the plant closing in 1971, the vertical gradient has been reduced by cessation of wastewater input. Further gradient reductions resulted from construction of the runoff collection ponds. The flux of contaminants vertically through the bog deposits is probably lower than it was during the last years of wastewater discharge to the bog. Short term variations in the rate of contaminant migration could result from influences such as fluctuations in precipitation, ground-water pumping or adjustments in the configuration of the agglomeration of free contaminants fluids in the saturated surficial sediments near well W13.

In summary, the available data indicate that the site is and has been a local source of contamination in the surficial ground water, and the nature and extent of the contamination is probably at a steady state for the near future. Appendix C describes the various available means of effecting remedial action to contain or reduce potential exposure to the contamination that has been characterized in this appendix. Appendix E establishes the nature and extent of reported contamination in the deep bedrock aquifers. Appendix D addresses potential contaminant transport via deep on-site multi-aquifer wells (wells W23 and W105). Ultimate discussions of site closure in these appendices and in the report reflect the nature and extent of contamination that has been described in this appendix.

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APPENDIX C
SITE REMEDIAL ACTIONS

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C1. INTRODUCTION AND OBJECTIVES

C1.1 Introduction

The soil at the former RT&CC plant site and surficial ground water both below and within 1400 feet east of the bog, which is adjacent to the south end of the former plant site, is contaminated with polynuclear aromatic hydrocarbons (PAH) and related compounds. In an attempt to determine what actions may be required to remedy this contamination, ERT has established objectives to be achieved by any remedial action, and using the criteria established by the National Contingency Plan (NCP), 40 CFR 300, ERT has screened a wide variety of available actions and examined in detail several of those actions.

There are two general categories of remedial actions available: (1) source-control actions and (2) off-site actions. Source-control actions are used to stop, reduce or otherwise control the migration or release of contaminants from the area at or near their original location. Off-site actions may be appropriate if contaminants have migrated beyond the area where they were originally located, and if it is necessary to mitigate and minimize a significant threat to the public health, welfare and the environment. This appendix deals with the various source-control remedial options available for the contamination at the site as described in Appendix B. Appendix D evaluates source-control options for well W23 and multi-aquifer wells. Appendix G evaluates off-site options.

The evaluation of source-control remedial action options was conducted using the cost-effectiveness criteria established by the NCP. These criteria include initial capital and operating costs as well as technical feasibilty. Specifically, the remedial action(s) sought is that which has the lowest cost, is technologically feasible, reliable, and is effective in mitigating and minimizing danger to the environment and in adequately protecting the public health and welfare.

C1.2 Objectives

ERT believes there are three distinct objectives that need to be satisfied to provide a comprehensive solution to the St. Louis Park problem, namely:

1) provide the City of St. Louis Park and neighboring communities with a safe and adequate supply of potable water at present and for the reasonably foreseeable future in a cost-effective and reliable manner;
2) allow for the present and reasonably foreseeable future uses of the ground-water resources in the St. Louis Park area in a cost-effective and reliable manner; and
3) allow for the safe, reasonable and beneficial use of the site for the reasonably foreseeable future in a cost-effective and reliable manner.

A detailed discussion of these objectives is presented in Section 3 of the report. As discussed in Section 3, source controls alone can not satisfy the first two objectives due to the extent of ground-water contamination. Source controls, however, may decrease (a) the time frame required to operate off-site controls or (b) the extent of off-site controls necessary, by minimizing potential contaminant migration from the site to aquifers.

The role of the site as a source of limited contamination to the Drift-Platteville is documented. This aquifer, however, is not a source of public water supply, and it is not likely to be a future source of any public drinking water. The goal of any remedial actions, therefore, is not to "clean up" this aquifer. This aquifer is, however, a potential conduit of contamination to lower bedrock aquifers, and remedial actions which control this conduit are evaluated in this appendix. Contamination of bedrock aquifers can not be directly attributed to the site based on the existing data.

A separate objective to be considered with respect to any remedial action considered for the site has been developed for this appendix. This objective is to minimize contaminant transport from the site to potentially impacted aquifers. A potentially impacted aquifer is an aquifer whose present or reasonably foreseeable future use as a public potable-water supply is affected by contaminant transport from the site. Source-control options which minimize contaminant transport to potentially impacted aquifers may (in conjunction with off-site controls) provide a cost-effective solution to the St. Louis Park problem.

The most direct benefit attributed to the implementation of source controls is to ensure the safe, reasonable and beneficial use of the site. The current uses of the site (i.e., residential and light industrial areas and a public park) are not affected by the contamination of the ground water and soil under and contiguous to the site and no remedial actions appear necessary. Future activities may include the extension of Louisiana Avenue through the bog area. These activities can proceed if future developers are aware of the site characteristics and if they take appropriate engineering and design precautions. Remedial actions such as site-use controls are considered in this appendix which will allow for the safe, future, and reasonably beneficial use of the site.

The specific objectives to be evaluated with regard to any remedial actions implemented at the site are:

1) Allow for the safe, reasonable and beneficial use of the site in the foreseeable future in a cost-effective and reliable manner.
2) Minimize contaminant transport from the site to potentially impacted aquifers.

It is important to note that the second objective is an indirect means of achieving the principal objectives discussed earlier in this section. Although a site source-control remedial action may be highly effective in terms of minimizing contaminant transport, implementing this source control may not be cost effective when compared to off-site controls which directly achieve the principal study objectives. This appendix evaluates all potential site source-controls which could meet the two objectives presented above. Integration of the source and off-site controls to meet the principal study objectives is discussed in Section 7 of the main report.

C2. SCREENING OF REMEDIAL ACTIONS

C2.1 Methodology

A wide variety of remedial actions are available to clean up or contain contamination at land disposal sites. Some of the methods are normally used in heavy construction projects such as excavation of soils and reburial in a secure landfill or construction of vertical barriers such as slurry walls that essentially prevent migration of contaminants. Other techniques are relatively new and experimental. For example, some wastes can be degraded by cultivating microbial activity in the contaminated soil.

The method utilized to screen applicable methods was to first identify the contaminated soil and ground-water zones and the various remedial action techniques. The applicability, advantages and disadvantages of each remedial action technique were described, and an initial judgement was made whether the technique was technically feasible considering site specific characteristics and the nature of the contamination. The most feasible remedial actions were then described in greater detail on a site specific basis considering such factors as design and construction considerations, effectiveness, and relative costs. Using these more detailed descriptions, each remedial action was evaluated considering the reliability, effectiveness, costs and the overall practicality of actual implementation. From this further screening, a combination of techniques was selected and recommended for in-depth evaluation as a viable option for implementation.

As a guideline and reference for the remedial action selection, the EPA manual entitled "Handbook, Remedial Action at Waste Disposal Sites," EPA-625/6-82-006, dated June 1982 was utilized.

C2.2 Contaminated Media Description

Previous studies completed by Hickok (1979) and Barr (1977) have identified the contaminated media for which remedial action is considered. These media, including soil and ground water, are described in detail in Appendix B.

When wastes are disposed of on land, certain constituents from the waste may migrate to air, water and earth resources. For purposes of recommending effective and practical remedial actions for the site, it is important to identify the contaminated media and to designate the categories of remedial action applicable to the site. From studies completed by Barr and Hickok, it is evident that both saturated and unsaturated inorganic and organic soils have been adversely impacted by PAH and related compounds.

For purposes of evaluating the applicability of the remedial action methods to the characteristics of the contaminated media, the four principal media are described briefly below:

bog deposits - composed of decomposed vegetation and varying amounts of peat, clay, silt or sand, and underlies filled land in the site. Most of these bog deposits have been covered by fill, with limited exposures in the southern section.

glacial drift - composed of thick stratified deposits of sand and gravel with silt and clay lenses, underlying the bog deposits. PAH and related compounds have locally penetrated these soils.

glacial-drift ground water - refers to the ground water that is contained in the glacial drift soil and has been reported to contain PAH and related compounds beneath the site and easterly from the site.

ground fluid - refers to a local lens of fluid organic material containing PAH and related compounds that occurs in the glacial drift, within approximately 100 feet of well W13.

These contaminated media are described in greater detail in Appendix B.

C2.3 Available Remedial Action Methods

C2.3.1 Introduction

For purposes of evaluating the various remedial action techniques, each method is placed within one of four classifications depending upon its overall function. The classifications are as follows:

removal - extraction of contaminated media from the site, including soil excavation and pumping of fluids.

containment - methods to keep PAH and related compounds from migrating off-site and to prevent physical contact with people.

in-situ treatment - methods used to treat or otherwise modify site contamination in place with and with minimal disturbance of existing conditions.

monitoring - measuring the physical and chemical changes in the site through ground-water analysis and evaluation, and site-use controls.

C2.3.2 Removal Methods

These methods involve either total excavation of the contaminated media or removal of contaminated water from the soil. The techniques are as follows:

- excavation of the contaminated fill, bog deposits, and underlying sand and gravel referred to as drift deposits. The excavated material would then be disposed of by transport to an off-site secure landfill, land treatment facility, incinerator, or a resource-recovery facility.
- removal of contaminated ground water by pumping special withdrawal wells followed by on-site treatment or discharge of the pumped ground-water to a sanitary sewer.

- removal of the ground fluid at well W13 by pumping installed withdrawal wells and separating the more dense oily, fluid from the pumped ground fluid, removing the separated oily fluid to a secure disposal facility and discharging the aqueous ground fluid to the sewer interceptor possibly after further treatment.

C2.3.3 Containment Methods

Containment methods pertain to a diverse group of construction procedures to prevent or otherwise limit the lateral and vertical migration of contaminants. These methods include the following:

- surface sealing or "capping" which involves placement and compaction of soils of low permeability over the contaminated media to minimize rainfall infiltration, leachate generation and human contact.
- vertical barriers such as slurry walls, steel sheeting, and grout curtains, installed to divert or impede ground-water flow. Slurry walls and grout curtains refer to a technique of inserting vertical sheets of expansive clay and/or cement concrete into the soil around the site.
- bottom liners which utilize an impermeable membrane at an appropriate elevation below or within a contaminated media to prevent vertical migration of contaminants.
- gradient-control wells which utilize one or more pumping wells to divert or otherwise modify the ground-water flow direction, by using withdrawal and/or injection wells.

C2.3.4 In-Situ Treatment

These techniques involve the use of various chemical compounds that are injected into the contaminated media or mixed with the media to stabilize or treat the PAH and related compounds. These methods include the following:

- bioreclamation, a treatment method where nutrients and oxygen are introduced into the contaminated ground water to stimulate the growth of microorganisms that degrade specific organic compounds.
- chemical injection, a method where chemical solutions are applied or injected into the contaminated media to leach out or stabilize the contaminated soil.
- encapsulation, a mixing and replacement technique that surrounds contaminated soil particles with a protective coating to prevent leaching.
- solidification, similar to encapsulation where the contaminated soil is excavated and blended with a chemical agent to solidify the mass and prevent leaching.

C2.3.5 Monitoring

Monitoring involves a variety of techniques to measure changes in soil, surface and ground-water quality. In addition, monitoring may include preventing unauthorized entry at certain locations, land-use restrictions, and measuring gas and vapor release. These activities, although not necessarily remedial actions, prevent inadvertent or intentional disruption of existing conditions, verify specific study conclusions and/or provide operating data for the implementation of specific remedial actions.

C2.4 Initial Screening of Remedial Actions

C2.4.1 Applicability to Site Characteristics and Contaminated Media

The selection of appropriate remedial actions for the site is based on an orderly review of the methods, the technology involved in implementation and the applicability to site characteristics and contaminated media. From this initial screening, a smaller group of methods are evaluated by introducing cost and implementation factors, and considering the effectiveness of the remedial action in meeting the objectives of the study.

Table C2-1 presents the remedial action methods screened, general advantages and disadvantages of each and an applicability rating of suitable or unsuitable.

For purposes of this study, the term "suitable" applies to techniques that are tried and proven, relatively simple to perform, and are effective in removing or otherwise controlling the migration of PAH and related compounds. Site specific factors which control the implementation of specific remedial actions were also considered in evaluating potential options. "Unsuitable" applies to techniques that will, in all probability, cause major site disruption with minimal benefits, be a complete misapplication of the technique considering site and contamination characteristics, or involve use of a technique which is at the research stage only.

C2.4.2 Potentially Suitable Remedial Actions

Table C2-2 summarizes the remedial actions evaluated in previous studies and those considered in this appendix. Off-site remedial actions including municipal water treatment, alternate drinking-water supplies, and multi-aquifer well closures are evaluated in Appendices D and G. Table C2-3 summarizes the suitability of the remedial actions evaluated and identifies the principal advantages or disadvantages of the techniques on which the technical judgement concerning suitability was based.

The remedial actions identified as potentially suitable are:

- excavation
- ground-fluid removal
- gradient-control wells
- bioreclamation
- monitoring/site-use control

As noted earlier, these remedial actions were selected based on technical feasibility relative to site specific factors and potential for meeting the study objectives. These objectives are (a) minimizing contaminant transport (as an indirect means of providing St. Louis

Park with an adequate, potable water supply and providing for the safe future use of the aquifers) and (b) allowing for the safe present and future use of the site.

The selected remedial actions are evaluated in detail in the subsequent sections considering the cost, technical feasibility and effectiveness of the remedial actions in meeting the stated objectives.

TABLE C2-1

INITIAL SCREENING OF REMEDIAL ACTIONS

Category	Method	Method Description	Advantages	Disadvantages	General Rating
Removal	Excavation, transport to off-site or on-site secure landfill	Total excavation of all waste and contaminated soil (above certain contaminant levels), loading and transport to licensed hazardous waste landfill constructed for this purpose. Excavated materials would be replaced by clean fill from off-site.	• removes source of contamination • unrestricted land use after removal • applicable to isolated and small pockets of waste and contaminated soil concentrations	• involves excavation adjacent to and beneath buildings, utilities and paved areas • excavation below water table requires waste water pumping, treatment and discharge • exposes waste to workers and public • off-site landfill will require hauling through urban and residential areas • excavation may cause embankment and structure instability • extensive amounts of soil refill needed • wet excavated material will require dewatering and extra handling prior to off-site shipment • if on-site landfill to be developed, extensive area needed with hazardous waste disposal permit • probably difficult to locate nearby secure landfill site to receive extensive quantities of contaminated materials • effectiveness will depend on prior and ongoing sampling and analysis.	Suitable
Removal	Excavation and land treatment	Excavation of waste and contaminated soil and transport to off-site land treatment facility. Refill excavated area with clean soil. Treatment is microbial degradation.	• effective for certain biodegradable wastes • removes source of contamination • no restricted use of site after removal • relatively simple technique	• not effective for non-biodegradable wastes • not generally applicable to buried wastes • requires transport through urban areas • exposure to workers • excavation below water table requires water pumping, treatment and discharge • excavation may cause severe disruption of existing roads, buildings, utilities • extensive amounts of soil refill needed • only effective during growing season	Unsuitable

Table C2-1 (Continued)

Category	Method	Method Description	Advantages	Disadvantages	General Rating
				• licensed land treatment site needed • extensive amounts of land area needed for treatment effectiveness will depend on prior and ongoing sample and analysis.	
Removal	Excavation and incineration	Excavation of wastes and contaminated soil and transport to off-site hazardous waste incinerator. Refill excavated area.	• removes sources of contamination • unrestricted use of site after removal • incineration destroys contaminants for all time	• wet waste would require some drying prior to shipment • inorganic portion of waste and soil would end up as ash, which may be the major portion • need to locate nearby hazardous waste incinerator capable of handling enormous quantities of very high ash content waste which may be non-existent • difficult to incinerate wet organic soils and soils with high percentage of sand • same excavation, structure stability and disrupted land use issues as other excavation options	Suitable
Removal	Ground water pumping	Pump out ground water by use of wells, pits or trenches to lower the water table, remove contaminated ground water and treat contaminated ground water.	• high degree of design flexibility, many techniques available • applicable to sites that are contaminating aquifers • minimal disruption of existing surface conditions • highly effective in permeable saturated soils	• indefinite continuous pumping with operation and maintenance costs • ground water extracted will require treatment or discharge to sewer system, surface water course, or seepage beds • may be very difficult to drain water from thick contaminated organic, and clayey soils • existing structure and embankment stabilization may be disrupted by lowering or altering water table • techniques are relatively new and innovative; may be considered experimental in some cases	Unsuitable
Removal	Ground Fluid withdrawal	Pump out fluids by use of extraction wells, separate fluid from water.	• removes potential source of contaminants in aquifer, minimal disruption of existing surface conditions • removes highly concentrated zone of contamination	• difficult to evaluate effectiveness • may be difficult to withdraw organic liquid from drift • disposal area or use needs to be found for organic fluid • wastewater from wells will require treatment and discharge.	Suitable

Table C2-1 (Continued)

Category	Method	Method Description	Advantages	Disadvantages	General Rating
Containment	Surface Sealing (capping) Includes grading, revegetation, surface water diversion	Regrading to improve surface runoff, diversion of surface water sources away from site area, placement and compaction of low to very low permeable soil with or without membrane barriers or soil amendments.	• reduces rainfall and snow melt infiltration into contaminated soil which reduces leaching of contaminants • reduced infiltration will tend to stabilize or depress water table thereby reducing contaminant migration • improves overall site appearance in uncovered contaminated swamps • provides physical separation between any exposed waste or contaminated soil and human or wildlife contact • very little exposure of wastes to workers	• leaching is reduced but not necessarily eliminated • can be easily disrupted by future uncontrolled construction activities • difficult to fill and grade around existing features such as underground utilities roads, buildings, and swamp • extensive amounts of soil cover would be needed from off-site sources • filling swamp surface may cause mud wave • flood storage capability in swamp may change existing drainage patterns	Unsuitable
Containment	Vertical Barriers	Sheet piling - driven interlocked steel sheet piling around perimeter for barrier.	• method is relatively easy • technique is a well established construction procedure	• probable structure damage from driving vibrations • high potential for initial leakage through interlocks • not effective in preventing vertical migration of contamination • sheeting would be very deep (50'+)	Unsuitable
		Slurry wall - perimeter trench excavation with placement of bentonite/cement/soil slurry.	• forms effective lateral barrier to ground-water flow	• not effective in preventing vertical migration of contaminants • very difficult site conditions for installations • difficult to seal coarse sand and gravel	Unsuitable
	Grout curtains	Pressure injection of chemical grouts to form barrier.	• minimal site surface disruption	• ground-water control needed inside barrier • may not be effective in organic soils • not suitable for diverting ground-water flow where gradient is nearly level • difficult to determine if barrier is continuous	Unsuitable

Table C2-1 (Continued)

Category	Method	Method Description	Advantages	Disadvantages	General Rating
Containment	Bottom liners	Injection of chemical grouts or installation of bottom liner below contaminated soil or waste to form a barrier to vertical leachate flow.	• effective in restricting vertical contaminated water flow	• not a feasible method for large sites where soil or waste would have to be removed and replaced • possible to install chemical grouts but effectiveness difficult to measure • very difficult location to perform work due to existing congested land use • leachate withdrawal system needed to prevent water table rise due to rainfall infiltration • difficult to measure effectiveness	Unsuitable
	Gradient Control Wells	Pump out ground water at variable depths and rates to contain the plume. Re-inject ground water or discharge to sanitary sewer.	• high degree of design flexibility • applicable to sites that are contaminating aquifers • minimal disruption of existing surface conditions • ground-water technology for method is readily available	• indefinite time period for pumping and equipment maintenance • water withdrawn may require treatment and discharge • structure stabilization and land subsidence possible from water table manipulation • active monitoring program needed to assess effectiveness • clogging of well screens possible	Suitable
In-situ Treatment	Bioreclamation	Injection of nutrients and air into ground water by use of injection wells.	• relatively fast and safe • effective for hydrocarbons • does not cause major water table changes with possible stability problems • minimal disruption of existing land surface	• long-term effectiveness is unsure • not effective for chlorides and solvents • effectiveness measurement requires considerable monitoring • expertise limited to a few companies, has not been applied to derivatives contamination by PAH and related compounds	Suitable
	Chemical Injection	Apply or inject chemical solutions, into contaminated soil or waste for neutralization. Contaminants are extracted or immobilized in place.	• minimal disturbance to existing site conditions • minimal exposure of waste to workers or public • technique is relatively simple	• high degree of uncertainty with respect to adequate waste contact • solution or by-product may be contaminant itself • difficult to determine effectiveness • ground-water withdrawal, treatment and discharge may be necessary for certain in-situ treatment methods • has not been applied to polynuclear aromatic hydrocarbon contamination	Unsuitable

Table C2-1 (Continued)

Category	Method	Method Description	Advantages	Disadvantages	General Rating
	Excavation and encapsulation then refill	Excavation and physical enclosure of soil or waste in synthetic encasement, then refill and cover	• generally suitable for small quantities of dry waste • completely isolates contaminants • encapsulated waste can be replaced into excavated area • improves land re-use potential	• not a tried and proven technology for SLP site characteristics and PAH contamination • waste must be dried • same excavation, structure stability and disrupted land use issues as other excavation options	Unsuitable
	Excavation and solidification, then refill	Excavation and blending of soil with chemical agents (cement bentonite, etc.), then refill and cover.	• generally suitable for small quantities of dry waste compatible with agents • improves land re-use potential	• organic soils with contaminants may not be compatible with agents • leaching of solidified product is possible • solidified waste cannot be redeposited below water table • generally applicable to specific dry wastes and not contaminated soils • same excavation, structure stability and disrupted land use issues as other excavation options	Unsuitable
Monitoring	Site security and monitoring	Limit access to site by fencing, gates, guards, and/or signs. Install ground and surface monitoring system and maintain.	• relatively easy to implement in specific locations • useful with other remedial action methods for health and safety protection	• does not mitigate sources of contamination • site surface is not a potential source of deleterious exposures	Suitable
	Land use restrictions	Designate certain areas for certain uses dependent upon existing conditions. Restrict land use in areas when waste is to remain.	• useful with other remedial action methods not including total waste removal • minimizes future liability problems for both structural and health purposes	• does not mitigate sources of contamination	Suitable
	Gas barriers and vents in structures	Installation of impermeable barriers, gravel-filled trenches and vents around structures to safely vent gas and vapors from buries waste.	• minimizes potential health and safety problems in and around structures overlying or adjacent to buried waste that is to remain in place	• suitable only for air quality in buildings • not applicable to ground-water contamination • difficult to determine needs • difficult installation in congested areas (buried utilities, pavements, structures)	Unsuitable

TABLE C2-2

IDENTIFICATION OF REMEDIAL ACTIONS PREVIOUSLY AND CURRENTLY UNDER CONSIDERATION

Remedial Action Method	Consultants Report & Date		
	Barr 1977	Hickok 1981	This Appendix
Ground Water Pumping	X		X
Excavation	X	X	X
Well Closures	X		
Additional Municipal Water Treatment	X		
Surface Sealing		X	X
Barrier-Slurry Walls		X	X
Barrier-Grout Curtains	X		X
Incineration (w/Excavation)		X	X
Encapsulation		X	X
In-Situ Treatment (Chemical Injection)		X	X
Solidification		X	X
No Action		X	
Ground Fluid Removal	X	X	X
Barrier-Sheet Piling			X
Bottom Liners			X
Gradient Control Wells	X	X	X
Bioreclamation			X
Site Security & Monitoring			X

TABLE C2-3

PRELIMINARY SCREENING OF POTENTIALLY APPLICABLE REMEDIAL ACTIONS

Remedial Action Category	Method	Suitability to Site Contamination	Comments
Removal	Excavation	Suitable	• well established technology, removes major portion of contamination
	Ground Water Pumping	Unsuitable	• impractical to lower water table below contamination, removal of contaminated ground water does not solve problem due to continued desorption
	Ground Fluid Withdrawal	Suitable	• removal can be easily implemented, removes highly concentrated source of ground-water contamination
Containment	Surface Sealing	Unsuitable	• ineffective in limiting contaminant transport
	Vertical Barriers	Unsuitable	• ineffective due to depth to bedrock
	Grout Curtains	Unsuitable	• ineffective due to ground-water gradient and presence of organic soils
	Bottom Liners	Unsuitable	• ineffective in limiting contaminant transport
	Gradient Control Wells	Suitable	• highly effective in limiting contaminant transport, can be easily implemented
In-situ Treatment	Bioreclamation	Suitable	• effective in degrading hydrocarbons
	Chemical Injection	Unsuitable	• not applicable to organics
	Encapsulation	Unsuitable	• impractical for large volume of contaminated media
	Solidification	Unsuitable	• ineffective for organics
Monitoring	Site Use Control	Suitable	• minimizes future liability problems
	Site monitoring	Suitable	• probably implemented with site use controls

C3. REMOVAL

C3.1 Excavation and Transportation to Off-Site Landfill

C3.1.1 Application

The initial screening of removal options indicates that the contaminated soil can be transported to an off-site secure landfill, or a constructed on-site secure landfill, land treatment facility, or to a hazardous waste incinerator. Due to the characteristics of the contaminated soil and the physical constraints based on preliminary volume determinations, the only disposal option for the excavated soils considered to be practical is the use of an existing secure landfill or development of a new secure off-site landfill. It is highly probable that one or more new secure landfill sites would be needed for the anticipated high volume of contaminated soil. Due to the very high proportion of natural inert soil grains to contaminated water or ground fluid, incineration and land treatment are not considered feasible disposal options for excavation. Most existing hazardous waste landfills would not be capable of accepting the anticipated volume of contaminated bog deposits, fill, and drift soils.

This method involves the physical removal of highly contaminated soil, loading, and shipping the material to an approved secure landfill for final disposal. The resultant space from the excavation would be refilled with inert soil.

The excavation method primarily applies to relatively dry soil or waste on small sites that can be easily delineated and excavated. By removal of contaminated soil, the leachate source is eliminated, and no new leachate would be generated. The method does not correct existing leachate-bearing ground water that has penetrated the aquifer, unless the leachate-bearing soil is removed also. Excavation of waste and contaminated soil requires a secure landfill for final disposal.

C3.1.2 Design and Construction Considerations

Excavation of dry solid waste in an above ground, man-made landform is relatively easy. At the site, however, the saturated waste and contaminated soil are mostly buried beneath fill. They lie near the surface only in a limited area in the bog. In many locations, contaminated soil underlies roads and buildings. Excavation of the contaminated soil below the water table would probably require a dragline and a backhoe operation.

Excavation of contaminated soil in the bog would create technical problems that would require extensive and sophisticated excavation and engineering techniques with high associated costs. The presence of existing structures such as Mill City Plywood and underground utilities will require demolition or relocation of the structures. Soil overlying the contaminated soil would require stockpiling or disposal. Excavation of the saturated contaminated soil would require stockpiling, draining and drying prior to loading onto sealed trucks which would transport the waste to a pre-designated secure landfill. The excavation will create wastewater (contaminated ground water from the excavation) that will require treatment. If this wastewater is not removed and treated, then the backfill soil will be exposed to contaminated ground water.

All of this activity may create severe slope and structure-stability problems that would require extensive shoring and sheeting. Exposure of contaminated soil and wastewater to workers would also occur in these operations. The resultant space from the excavation would require backfilling by an equal volume of clean soil plus an additional amount to compensate for settling. Backfilling is necessary to restore the site area to its original ground surface grade.

A wide variety of very difficult and costly implementation problems characterize the total excavation method, and include the relocation of businesses and transportation routes. A limited application of this technique may be appropriate for the removal of a portion of the contaminated bog deposits and inorganic soil in the swamp area where locally elevated concentrations of contaminants have been detected. The following subsections describe the procedure that would be necessary for total excavation. Figure C3-1 indicates the location of each area described.

C3.1.3 Procedures for Total Removal of Contaminated Soil and Waste

Area 1

1. Excavate an estimated quantity of 98,000 cubic yards to remove the existing soil stockpile (located in the southwest corner of the former plant site), load and transport to a licensed secure landfill, if testing indicates contamination.
2. Excavate filled area (approximately 27 acres) to a depth of up to 20 feet to remove contaminated soil. The anticipated volume of material is 872,000 cubic yards. Excavation could be accomplished by front end loader to an elevation close to the water table, and the remainder would be excavated by dragline. The material below the water table is saturated and would require some draining and drying prior to loading. The excavated material would then be shipped to a secure landfill. The water remaining in the excavation would be contaminated and would require pumping to an on-site treatment plan or to a sanitary sewer for off-site treatment.
3. The existing runoff retention pond, when drained, is assumed to contain up to 2 feet of contaminated soil in the bottom. Removal of 10,000 cubic yards of material in the bottom over 3 acres will probably require a dragline operation with drainage drying, loading and transport to a secure landfill.
4. Refill of the total excavated area of 30 acres will require approximately 1,162,000 cubic yards of clean soil backfill, including an allowance of 20% for compaction shrinkage. This material would be hauled to the site from off-site borrow pits.

NON-RESPONSIVE

[redacted]

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5. Following refill, the total area of 30 acres will require loaming, seeding, or paving as appropriate.

Area 2

1. Perform subsurface explorations to determine if contaminated peat and soil exist beneath Walker St., two buildings, and Highway 7. Assuming that significant amounts of contaminated soil exist and the material is to be excavated, the following work items would need to be carried out. These work items do not include the legal factors and socio-economic impacts attendant to disruption of existing land use.
2. Close an 1100 foot long section of Walker St. and remove all buried and overhead utilities and relocate. Remove pavement and underlying clean fill and stockpile nearby. Excavate contaminated soil to and below the water table as needed. Excavation would probably be accomplished by front end loader to the water table and dragline below the water table. Saturated contaminated soil would be drained prior to loading on to sealed trucks. All contaminated soil would then be transported to a secure off-site landfill. It is anticipated that the maximum depth of soil removal is 25 feet and would total approximately 41,000 cubic yards. Ground water encountered in the excavation would be considered contaminated. It would be pumped out, treated and discharged off-site or pumped directly to a sanitary sewer interceptor. The stockpiled roadway fill can then be used as refill with the addition of about 51,000 cubic yards of clean granular fill from off-site. The utilities may then be replaced and roadways and bridges reconstructed.
3. For removal of contaminated soil beneath the two structures on site (Mobile Marine Discount and Mill City Plywood), totaling 1.6 acres, it will be necessary to remove or demolish the structures, excavate the foundations, and remove all buried and overhead utilities prior to excavating the contaminated soil. The excavation would be performed in

conjunction with the excavation of the contaminated soil in the adjacent filled area. If the structures are on pile foundations, the extraction of contaminated soil will be extremely difficult.

4. Excavation of the contaminated soil in the remaining bog encompasses an area of approximately 2.2 acres. It is estimated that if an average thickness of 15 feet of bog deposits is to be removed, then a total of 53,200 cubic yards will need to be excavated by dragline, drained and transported to a secure landfill. If an average thickness of 10 feet of contaminated Middle Drift granular soil underlies the bog deposits, then an additional 36,000 cubic yards of soil will need to be removed, drained, and transported to a secure landfill. Contaminated water in the resultant excavated area will need to be pumped out for on-site treatment and discharge or piping to a sanitary sewer interceptor. The refill operation would be more or less continuous with the excavation and would require at least 102,000 cubic yards of clean granular fill.

5. Excavation of contaminated Middle Drift soil and bog deposits beneath existing filled areas, including the building area, involves approximately 6.5 acres. Work would include excavation and stockpiling of existing clean fill to expose the contaminated bog deposits and soil. This stockpiling activity would be followed by a front end loader and dragline operation to remove an estimated amount of 157,000 cubic yards of bog deposits and 105,000 cubic yards of contaminated soil to an average depth of 25 feet. The saturated excavated material will require draining prior to loading and shipment to a secure landfill. The contaminated water within the excavation will require pumping and on-site treatment or piping to a sewer interceptor. Refill of the area will consist of the purchase and placement of about 301,000 cubic yards of clean granular fill. Building, paving or revegetation may then be accomplished along with the installation of necessary utility lines.

6. An existing 52 inch drainline from the drainage pond in Area 1 passing through Area 2 will require removal and relocation as part of the overall project to remove any contaminated soil beneath the line. It is anticipated that about 700 linear feet of drainline will be involved in Areas 1 and 2.

7. Removal of contaminated soil beneath Highway 7 would require the closing of an 1100-foot length of highway, and temporary diversion of traffic. The removal operation would involve pavement, subgrade, highway embankments, fencing signs, etc., prior to reaching the surface of the contaminated Middle Drift granular soil and bog deposits. Excavation of bog deposits for an average of 15 feet in thickness would be about 61,000 cubic yards. Removal of 10 feet in thickness of contaminated soil beneath the peat would be about 41,000 cubic yards. The same excavation, draining, transport, and water treatment procedures pertain to this section as the others previously described. If the bog deposits had been removed or are thoroughly surcharged resulting from highway construction, then excavation of the Highway 7 corridor may not be warranted. If a total depth of 25 feet of soil and peat is excavated from beneath the roadway, then at least 116,000 cubic yards of clean granular fill would be required for refill. The road would have to be reconstructed following refill.

Area 3

1. Excavate contaminated peat from the bog consisting of about 3 acres, with an average excavation depth of 15 feet. It is estimated that approximately 72,000 cubic yards of saturated bog deposits would be removed, drained, and shipped to a secure landfill site.

2. Relocate a 500-foot long section of the 52 inch pipeline traversing beneath site in order to remove contaminated bog deposits and Middle Drift granular soil beneath.

3. Relocate a 500-foot long section of Lake St. (0.6 acres) if it is found that contaminated bog deposits and Middle Drift granular soil underlies the road.
4. Excavate the filled area of approximately 8 acres to remove contaminated Middle Drift granular soil and bog deposits. Using an average thickness of 10 feet of soil, the total to be excavated is 130,000 cubic yards. To remove the underlying bog deposits, a quantity of 194,000 cubic yards is estimated using an average thickness of 15 feet. Contaminated material excavated below the water table will require draining prior to shipment. Water in the excavation will require pumping and treatment.
5. Refill the entire 11 acres with clean granular fill from off-site sources. It is estimated that 84,000 cubic yards will be needed to refill the swamp to its original grade, and 373,000 cubic yards to refill the filled areas to current grade, for a refill total of 457,000 cubic yards. The final graded surface would then be revegetated or used for other purposes. The relocated Lake St. would then be restored to its original position.

C3.1.4 Reliability and Effectiveness

The excavation method is generally considered to be one of the most reliable and effective methods for eliminating a small, well defined source of ground-water contamination. Total removal of all contaminated soil is a direct remedial action best applied to relatively small, shallow, well defined sites and contaminant sources. However, the site covers an extensive land area, is surrounded by intensively used land, and has indistinct contamination boundaries. The site is not a suitable candidate for total removal of all contaminated soils.

The effects of sorption/desorption phenomena are not well understood and it is not possible to identify whether there would be any significant benefit from partial removal of select areas of highly contaminated soil. For instance, the bog deposits contain some of the highest concentrations of benzene extractables at the site. Available data, however, indicate that the organic matter content of soil significantly affects the sorption/desorption of specific organic contaminants (Lambert et al. 1965, Karickhoff et al. 1979). Removal of the most highly contaminated bog deposits may not significantly reduce contaminant transport due to the high sorptive capacity of organic soils. The available data suggest that partial removal of contaminated soil will not be effective in minimizing contaminant transport due to the difficulty in identifying (a) the most significant source of surficial ground-water contamination and (b) the location of the most significant contaminant source. Determination of the location and magnitude of contamination source would require extensive laboratory studies and field exploration programs and would most likely yield limited results.

C3.1.5 Costs

Costs for implementation of the excavation option are particularly difficult to develop due to the wide variety of surface and subsurface conditions that influence cost estimating. It would be necessary to perform an extensive engineering, environmental and socioeconomic study to prepare accurate cost estimates. The intensity of land use, especially the presence of roadways and structures in the site area, may lead to excessive and unacceptable costs and community disruption.

Table C3-1 presents the estimated quantities involved in excavating contaminated soil. For purposes of developing approximate quantities, an average thickness of 15 feet of peat and 10 feet of underlying Middle Drift deposits has been utilized. The total estimated quantity of contaminated peat Middle Drift deposits is 1,933,000 cubic yards. This material would have to be shipped to an existing or newly constructed hazardous waste landfill. In order to refill the excavated areas, approximately 2,148,000 cubic yard of clean granular fill will be needed from off-site sources.

Tables C3-2 and C3-2A present the estimated costs for major components of the excavation options. Costs listed are those that can be estimated with a reasonable level of accuracy. Many other

TABLE C3-1

ESTIMATED QUANTITIES FOR EXCAVATION OPTION

Area	Location	Acreage	Removal Depth in Feet	Volume of Excavated Material Cubic Yards	Volume of Refill Cubic Yards
1.	Soil Stockpile	-	-	98,000	
	Filled area	27	20	972,000	1,162,000
	Runoff Control Pond	3.0	2	10,000	-
2.	Walker St.	1.0	25	41,000	51,000
	Bog	2.2	15	53,000	61,000
	Filled Areas/Structures	6.5	25	262,000	301,000
	Highway 7	2.5	25	101,000	116,000
3.	Bog	3.0	15	72,000	84,000
	Filled Areas	8.0	25	324,000	373,000
	TOTALS			1,933,000	2,148,000

NOTE: All quantities are estimates based on utilization of stereo aerial photographs and are for planning purposes only.

TABLE C3-2

PARTIAL ESTIMATED COSTS FOR TOTAL EXCAVATION

Work Item	Total Estimated Cost Range
1. Total Excavation of Areas 1-3	$ 11,600,000 - 24,000,000
2. Preparation and Loading for Shipment	4,000,000 - 7,000,000
3. Material Transportation to 100 miles	29,000,000 - 39,000,000
4. Disposal Cost for Bulk Solids at Landfill	145,000,000 - 242,000,000
5. Refill of Excavated Areas	15,000,000 - 21,000,000
6. Revegetation of Refilled Land	71,000 - 142,000
Total*	$204,671,000 - 333,142,000

*Cost Items excluded from total are as follows:

- structure removal & replacement, including foundations
- roadway removal & relocation
- wastewater pumping, treatment & disposal (contaminated groundwater)
- sheeting and shoring for perimeter stabilization during excavation
- underground utility excavation, removal and relocation
- overhead utility relocation
- work activity monitoring
- project engineering & design
- subsurface investigations
- owner compensation for disrupted and/or structure use
- waste material transportation costs beyond 100 miles from site

TABLE C3-2A
COST BREAKDOWN BY ACTIVITY*

Work Item		Total Estimated Cost Range
1.	Excavation	$ 11,600,000 - 24,000,000
2,3,4.	Disposal	178,000,000 - 288,000,000
5,6.	Refill (backfilling)	15,071,000 - 21,142,000

associated work items cannot be properly estimated at this time, and these are listed separately. The tables demonstrate the extremely high costs for the total excavation option even without including the work items that are particularly difficult to estimate.

The costs also assume that a suitable hazardous waste landfill will be able to accept the extensive quantities of material. The secure landfill is assumed to be not farther than 100 miles from the site.

Tables C3-3 and C3-3A present partial estimated costs for excavations as shown on Table C3-2. Area 2 has been identified as a location where elevated levels of contaminant are significantly higher than in Areas 1 and 3.

C3.1.6 Advantages and Disadvantages

An evaluation of the total excavation method reveals distinct advantages and disadvantages. The two categories are listed separately below for purposes of comparison.

<u>Advantages</u>

- Removes a source of contamination for all time;
- provides generally unrestricted land use after completion; and
- is beneficial for small, well identified waste or contaminated soil deposits.

<u>Disadvantages</u>

- Involves excavation adjacent to and beneath buildings, utilities, and paved areas;
- excavation below water table will require water pumping, treatment and appropriate discharge;
- excavation exposes waste to workers and general public during transport;
- excavation may cause embankment and structure instability;

TABLE C3-3

PARTIAL ESTIMATED COSTS FOR EXCAVATION* OF AREA 2 ONLY

	Work Item	Total Estimated Cost Range
1.	Excavation	$ 2,742,000 - 5,484,000
2.	Preparation and Loading for Shipment	915,000 - 1,660,000
3.	Material Transportation to 100 Miles	6,855,000 - 9,140,000
4.	Disposal Cost for Bulk Solids at Landfill	34,275,000 - 57,125,000
5.	Refill of Excavated Area	$ 3,703,000 - 5,290,000
6.	Revegetation	12,000 - 24,000
	Total*	$48,502,000 - 78,723,000

*Cost items excluded from total include those indicated in Table C3-2.

TABLE C3-3A

COST BREAKDOWN BY ACTIVITY*

	Work Item	Total Estimated Cost Range
1.	Excavation	$ 2,742,000 - 5,484,000
2,3,4.	Disposal	42,045,000 - 67,925,000
5,6.	Refill (backfilling)	3,715,000 - 5,314,000

- extensive amounts of clean soil backfill are needed, and nearby sources may be limited;
- saturated contaminated Middle Drift granular soil and bog deposits removed below the water table will require extra handling, conditioning or drying prior to off-site shipment;
- in all probability, it will be extremely difficult to locate an existing secure landfill or establish a new site that would be capable of receiving up to two million cubic yards of contaminated material;
- estimated costs for total removal are exceptionally high and unpredictable, based on total land use disruption and reclamation;
- even ignoring disposal costs, the excavation is a significant cost;
- requires continued analysis of soil samples to identify limits of excavation;
- removal of roads, utilities and structures, surface water courses and subsequent relocation presents social, environmental and economic problems requiring independent study;
- extensive subsurface exploration program is needed to define the limits of excavation, including the determination of the nature of contaminated soil beneath structures, roads and paved areas;
- the issue of owner compensation for disrupted land or structure use is highly unpredictable; and
- if no landfill sites are available nearby, the costs will be significantly higher.

C3.2 Ground-Fluid Removal and Disposal

C3.2.1 Application

The removal of the ground fluid and subsequent disposal refers to the ground fluid encountered by the pumping of well W13 between Highway 7 and Lake Street. The fluid was encountered about 50 feet below the ground surface in the Middle Drift (Erlich 1982). The relatively high concentrations of contaminants in the ground fluid may act as a continuing source of contamination to the Drift-Platteville aquifer. The removal would be carried out by the installation of a ground-water withdrawal well or wells.

C3.2.2 Design and Construction Considerations

Of prime consideration in withdrawing the ground fluid is defining its vertical and horizontal limits. This would be achieved by performing a detailed soil boring and soil and water sampling program in the well W13 vicinity. When the limits and characteristics of the ground fluid are defined, well design and treatment disposal options can be determined.

At present, the only data available on the ground fluid are from well W13. Analyses of the combined two phase fluid which comprises the ground fluid and the aqueous phase associated with the ground fluid are presented in Table C3-4. Detailed analyses of the ground fluid reveal that it is approximately 0.5% oil and grease and is characterized by very high concentrations of total PAH (970 mg/l) (Erlich et al. 1982).

The volume of the ground fluid has been estimated in Appendix B. As noted in Appendix B, the extent of the ground fluid can be bounded by boring and well quality data in the vicinity of well W13 and is probably less than 100,000 gallons. Minimum volumes are not known, although it is possible that the ground fluid is restricted to a very small area around well W13. For the purpose of this evaluation, volumes of 10,000 and 100,000 gallons are assumed. There are no data to suggest that other ground fluid bodies exist. Considering the

TABLE C3-4

CHARACTERISTICS OF THE FLUID

WITHDRAWN FROM WELL W13

	Total							Aqueous Phase		Fluid Phase
Date Sampled	2/18/77	5/26/77	5/26/77	6/2/77	6/22/77	3/29/79	3/29/79	7/80	2/81	1980
Phenolics (mg/l)	4.8	51	56	49	50	27	81	29	29.1	
Total Organic Carbon (mg/l)	1,171					6,900	6,000	150		
Chemical Oxygen Demand (mg/l)				25,800	40,200					
Total Dissolved Solids (mg/l)				1,617				1,400		
Specific Conductance (mhos/cm @ 25°C)						2,000				
Arsenic (mg/l)			0.035			0.027				
Cadmium (mg/l)				0.01						
Copper (mg/l)				0.05		0.003				
Lead (mg/l)				0.01		0.004				
Zinc (mg/l)				0.09						
pH				7.6	7.5	7.6				
Sodium (Tot. Recov.) (mg/l)						370		430	430	
Chloride, Dissolved (mg/l)						250		303		
Sulfate Dissolved (mg/l)						52				
Ammonia Nitrogen (Total)(mg/l)						6.9				
PAH (total) (mg/l)									22.3	970
Oil and Grease (mg/l)					4,900					
Data Source	BARR 1977	BARR 1977	BARR 1977	BARR 1977	BARR 1977	Hult 1981	Hult 1981	Ehrlich 1982	Ehrlich 1982	Ehrlich 1982

large number of borings that have been placed in the site area, the possibility of other ground fluid bodies existing appears minimal.

Removal of the ground fluid by pumping would be expected to leave a significant amount of PAH and related compounds adsorbed to the soil which is in contact with the ground fluid. In addition, removal of the ground fluid may result in an increased net surface area of contaminated drift deposits exposed to the ground-water flux. It is hypothesized that while the ground fluid is in place, only the periphery of the ground fluid is exposed to the ground-water flux (i.e., no ground water flows through the ground fluid). The soil which had been surrounded by the ground fluid, therefore, would be exposed to the ground-water flux after removing the ground fluid. Thus, the net effect of pumping the ground fluid may be to temporarily increase the concentration of PAH and related compounds in the Middle Drift ground water.

A gradient-control well system could be used to minimize contaminant transport if the ground fluid were pumped and the pumping resulted in a temporary increase in the concentration of PAH and related compounds in the Middle Drift ground water. A detailed description of a gradient control well system which could meet this objective is presented in Section C4.

Disposal of the recovered ground fluid is assumed to be to the municipal sanitary sewer system. It is assumed that the oil and grease in the ground fluid will be separated by a gravity separation unit prior to discharge and the collected oil and grease will be disposed of off site as a hazardous waste by a licensed contractor. It is anticipated that a typical well will be about 50 feet deep, at least 6 inches in diameter, and able to pump at variable rates from 10 to 25 gallons per minute. It would take less than one month to pump 10,000 to 100,000 gallons at a pumping rate of 10 gallons per minute.

The design must consider the heavier-than-water characteristics of the oily phase and its high viscosity which will have a tendency to clog conventional gravel packs and screens. A determination must also be made relative to the type of material used in the well, since disintegration and/or encrustation may occur. A typical well may have a 4 inch submersible pump to withdraw the ground fluid which will also pump ground water. Frequent monitoring will be required to determine the effectiveness of the pumping. The monitoring will permit estimates to be made regarding the length of time necessary to operate the system.

Other considerations for implementing the ground-fluid removal system include disposal of the collected organic fluid, potential interference with existing land use, location of overhead and underground utilities, preparation of a detailed operation plan, and treatability of the collected organic fluid and ground water.

C3.2.3 Reliability and Effectiveness

The effectiveness of removing the ground fluid in terms of reducing contaminant transport must be evaluated in terms of the relative contribution of the ground fluid as a source of PAH and related compounds to the Drift ground water.

Based on a maximum volume of 100,000 gallons and a PAH concentration of 970 milligrams per liter, the total mass of PAH present in the ground fluid is approximately 800 pounds. The concentration of PAH present in the aqueous phase of the ground fluid is 22.3 milligrams per liter (Erlich et al. 1982). The ground fluid is composed of 0.5% oil and grease (in other words 99.5% water) so the total mass of PAH present in the aqueous phase of the ground fluid is assumed to be approximately 20 pounds. Ground-water flow velocities in the Drift-Platteville are estimated to be 0.5 feet per day and the base of the ground fluid (parallel to the direction of ground-water flow) is assumed to be 200 feet long. Thus, ground-water travel times are on the order of 1 year in the vicinity of the ground fluid. The above assumptions suggest that the relative contribution of the ground fluid is on the order of 20 pounds of PAH per year to the drift ground water.

Field data suggest that the impact of the ground fluid is strongly localized on Drift ground-water quality in the vicinity of well W13. It is not possible to determine whether the ground fluid is a significant source of Drift ground-water contamination relative to the role of the contaminated bog and Drift deposits.

Removal of the ground fluid may result in a measurable improvement in ground-water quality in the vicinity of well W13; however, it may also result in a temporary flush of PAH and related compounds from the soil in contact with the ground fluid. Removal of the ground fluid is expected to result in no impact on ground-water quality in the St. Peter or deeper aquifers due to the long hydraulic travel times and effects of degradation, dispersion and retardation. Furthermore, removal of the ground fluid does not remove the contaminants present in the Drift-Platteville ground water or the contaminated soil.

The available data suggest that:

- Removing the ground fluid is moderately effective in minimizing local contaminant transport in the Drift aquifer.
- Removing the ground fluid will not result in a measurable benefit with regards to meeting the three principal remedial action objectives for the site.

C3.2.4 Costs

Costs for the ground-fluid removal are presented in Table C3-5 and include treatment costs. All costs are assumed to be capital costs due to the short time frame involved.

C3.2.5 Advantages and Disadvantages

Consideration of the various factors involved in the design and development of the ground-fluid removal system resulted in the formulation of a series of advantages and disadvantages listed below:

<u>Advantages</u>

- Technique has high degree of flexibility relative to location, depth, well design, pumping rates, disposal options;
- minimal disruption of existing land use in comparison to other techniques such as excavation or vertical barriers;

TABLE C3-5

COSTS FOR GROUND-FLUID RECOVERY

	Capital
Exploration	10,000
Analysis & Reporting	15,000
Production Wells (2)	20,000
Ground-Fluid Treatment	
Sewer Hook up	10,000
Separation Unit	10,000
Sewer Discharge Fee ($.70/1000 gallons)	7,000
Off Site Disposal (5000 gallons)	5,000
Operation and Maintenance	
1 Technician/30 days ($24/hour)	6,000
Monitoring (10 samples, $200/sample)	2,000
TOTAL	85,000

- the technology for ground water and other fluid withdrawal systems is well established;
- probability of land subsidence is low if wells are spaced apart sufficiently;
- costs for well installation are generally lower than for other remedial actions;
- removes a known source of contamination to Drift aquifer; and
- reduces uncertainties associated with potential future impacts.

Disadvantages

- Difficult to evaluate effectiveness of removal upon completion of entire aquifer;
- high viscosity organic fluid may clog well screen and prevent effective pumping;
- organic fluid collected will require acceptable disposal procedure;
- ground water extracted will probably require discharge line to sanitary sewer and subsequent treatment;
- removal of the ground fluid may result in temporary flush of contaminants to Drift aquifer; and
- may require a gradient-control well system to control flush of contaminants.

C4. CONTAINMENT

Section C2 presented the range of potential options for containing the contaminated source materials. The only containment option which appears technically feasible is gradient-control wells due to the extent of the contaminated bog and Middle Drift deposits as well as site specific constraints which affect the implementation of containment options.

C4.1 Application

Implementation of a gradient-control well system is applicable to all contaminated media at the site (i.e., soil, ground fluid and ground water). The gradient control system is designed to control the movement of ground water which comes in contact with the contaminated bog deposits, Middle Drift deposits and the ground fluid. This ground water can potentially become contaminated and act as a continuing source of contamination to the Drift-Platteville aquifer. The gradient-control well system, however, is not designed to contain and remove the entire plume of contaminated ground-water in the Drift-Platteville which has already moved beyond the site.

Operation of a gradient-control well system in the Middle Drift may eventually remove most of the contamination present in the bog and Middle Drift deposits through desorption mechanisms. The time frame however is probably in the order of hundreds of years (Barr 1977).

C4.2 Design and Operating Considerations

The principal design considerations associated with the gradient-control system are the pumping rate and placement of wells. A methodology for developing a conceptual design of a gradient control well system is presented in Lundy and Mahan (1982). The plume discharge can be calculated by:

1) $Q = TIW$

where

Q = Volumetric discharge through a part of an aquifer that encompases the plume

T = Aquifer transmissivity

I = Hydraulic gradient across plume

W = maximum width of plume measured at right angles to gradient

A well pumping rate equal to the volumetric discharge (Q) will induce all water within the plume to move towards the well. The well will capture ground water within a distance downgradient (x_o) from the well defined by:

2) $x_o = \frac{-Q}{2\pi TI}$

The plume boundary can be calculated on a cartesian (x,y) coordinate system where y is the width of the plume at a distance x from a pumping center by:

3) $y = -x \tan \left(\frac{2\pi\ TI}{Q} y\right)$

As noted in Section C4.1, the purpose of the gradient-control well system is not to contain a plume of contaminated ground water but to capture the ground water which comes in contact with the contaminated ground fluid, bog and Middle Drift deposits. The width considered in equation 3, therefore, is the maximum extent of contaminated source materials taken perpendicular to the gradient.

As noted in Appendix E, this approach is only valid if there is no significant recharge or withdrawal to the aquifer. Although it is known that there are many private wells screened in this aquifer, their effect is not shown in any field potentiometric measurements. The wells are assumed to result in a negative recharge which balances any recharge due to rainfall infiltration.

Aquifer transmissivity and gradient significantly affect the selection of a pumping rate. Aquifer transmisivities of 7,600 to 31,300 gallons per day per foot have been estimated for the Middle Drift (Hickok 1979). The ground-water gradient is assumed to be directed due east at 12 feet per mile (Hult and Schoenberg 1981).

Figures C4-1, C4-2 and C4-3 present the capture area for a well located downgradient from the site at transmissivity values of 10,000, 20,000 and 30,000 gallons per day per foot and a well pumping rate of 100 gallons per minute and 50 gallons per minute. The figures present the location of selected monitoring wells in the Middle Drift and Platteville for which water quality data are available and depicts the contamination status of these wells based on the background phenolics presented in Appendix B.

A pumping rate of 100 gallons per minute is adequate to contain the area of <u>known</u> ground-water contamination which is in contact with the contaminated source material for all the transmissivity values considered. It should be noted however that:

- the data are inadequate to define the downgradient (eastward) extent of ground-water contamination by PAH and related compounds, and
- if a transmissivity value of 30,000 gallons per day per foot is assumed, a pumping rate of 100 gallons per minute does not contain all ground water which comes in contact with the contaminated bog and Middle Drift deposits.

As discussed in Appendix B, the contaminated drift deposits occur in a few local instances, discontinuously down to the Drift-Platteville contact, although contaminated ground water exists directly below the

NON-RESPONSIVE

[redacted]

NON-RESPONSIVE

[redacted]

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NON-RESPONSIVE

[redacted]

site in the Platteville shown on the map (W22 and W26). Due to the unknown pattern of solution channels which control ground-water flow in the Platteville, it will be most effective to manage gradients in the Middle Drift. As the piezometric levels are lowered in the Middle Drift, a portion of the flow in the Platteville will be diverted up through the Lower Drift and captured by the gradient control wells (Barr 1977).

The gradient-control wells should be located near the eastern extent of the contaminated bog and Middle Drift deposits (see Figure C4-1). This will contain the ground water which comes in contact with the contaminated deposits as well as most of the contaminated plume which is now located between the site and the tributary buried bedrock valley. In addition, a gradient control well in the vicinity of well W13 should be operated to control leakage of the highly contaminated ground water in this area into the Platteville.

The primary concern associated with contaminants present in the Drift-Platteville is the hydraulic connections to the St. Peter through the tributary bedrock valley and multi-aquifer wells. Modeling indicates the hydraulic gradient in the St. Peter is directed to the east in the area of these connections and would transport potential contaminants away from the only drinking-water supply well (SLP3) screened in the St. Peter. Available water-quality data support this conclusion and indicate that contamination has only been detected in the vicinity of two multi-aquifer wells (W27 and W33) (Barr 1977). W33 has been reconstructed and is now only open to the St. Peter. The available data, therefore, indicate that at this point in time it is premature to assume that water quality in well SLP3 is or will be affected by contaminated Drift-Platteville ground water.

The gradient-control well(s) will not provide complete capture of all ground water which comes in contact with the contaminated media. Significant leakage effects to the Platteville and St. Peter may occur by:

- vertical leakage from the Middle Drift to the Platteville,
- horizontal leakage from the Platteville and Middle Drift into the buried bedrock valley and subsequently into the St. Peter,

- vertical leakage from the Platteville through the Glenwood into the St. Peter, and
- leakage through multi-aquifer wells which connect the Middle Drift and Platteville with the St. Peter.

Placing the gradient-control well(s) as close to the zone of contamination as possible will minimize these leakage effects. The following summarizes the capabilities of a gradient-control well(s) in the Middle Drift with respect to limiting off-site migration of contaminated ground water:

- Lateral flow of ground water in the Middle Drift which has come in contact with the contaminated source material will be intercepted by the gradient control well(s).
- Multi-aquifer wells downgradient from the gradient-control wells will not intercept Drift ground water which has come in contact with the contaminated source material.
- The area over which leakage of potentially contaminated ground water can occur (from the Drift to the Platteville) will be restricted to the site area.
- Lateral flow of ground water in the Platteville will be partially intercepted by gradient-control wells in the Middle Drift.

The present data base is inadequate to develop a final design for the gradient-control well system. Data regarding ground-water quality and movement in the Middle Drift and Platteville which have been collected by the USGS and are not presented in Hult and Schoenberg (1981) or Ehrlich et. al. (1982) should be integrated into the analysis prior to developing a final design.

The present data base is adequate to define a conceptual design for the system. The following points highlight the principal criteria used for evaluating the economic feasibility of the system:

- well(s) should be screened in the Middle Drift,

- combined pumping rate of all well(s) is approximately 50 to 100 gallons per minute,
- collected ground water will be discharged to a sanitary sewer (see Appendix F), and
- well(s) should be located at the eastern extent of the contaminated bog and drift deposits (see Figure C4-1) with the exception of a well located in the vicinity of well W13.

Design of a gradient-control well system should provide sufficient flexibility to increase or decrease pumping rates as appropriate. It is assumed that the gradient-control well system will include detailed ground-water monitoring of the Drift and Platteville to identify the effectiveness of the system. Monitoring may indicate that additional wells or higher pumping rates are necessary to contain the ground water which comes in contact with the contaminated bog and Drift deposits.

Operation of the gradient-control well system is assumed to be 100 years. Monitoring, however, may indicate that the contaminated bog and drift deposits effect on ground-water quality significantly diminishes with time. Treatment requirements for the collected ground water may, therefore, decrease or be eliminated with time. Treatment and disposal options for the collected ground water are discussed in Appendix F.

C4.3 Reliability and Effectiveness

Gradient-control well systems are a well established reliable technology for the containment of contaminated ground water (EPA 1982). The effectiveness of the system is dependent on the aquifer characteristics. As noted earlier, leakage from the Middle Drift may occur under the proposed system. Leakage could be decreased by also developing a gradient-control well system in the Platteville. The poorly understood pattern and hydraulic characteristics of solution channels in the Platteville, however, create considerable uncertainties regarding the placement and effectiveness of recovery wells in this bedrock unit.

As noted earlier, the connection between the contaminated ground-water in the Drift-Platteville to ground-water quality in the vicinity of St. Louis Park production wells has not been established. Modeling results presented in Appendix E indicate that contaminated ground water in the Drift-Platteville aquifer does not and will not impact present potable water supplies in any of the bedrock aquifers except under extreme pumping condition (for well SLP3). Available water-quality data support this conclusion.

Several uncertainties exist, however, concerning the mechanism of contaminant transport from the Drift-Platteville to lower bedrock aquifers. The role of multi-aquifer wells as conduits for the transport of contaminated Drift-Platteville ground water is imperfectly understood. Although contaminant transport to lower bedrock aquifers via bedrock valleys downgradient from the site has not been documented, they are potential conduits by virtue of mapped ground-water flow paths. Implementation of the gradient control well system will limit any uncertainties associated with the impact of these pathways on potentially affected aquifers.

C4.4 Costs

The estimated cost for installing a gradient-control well system is dependent on the number of wells installed and the discharge method selected. For cost estimating purposes it is assumed that:

- two 6-inch diameter wells are installed to a depth of 50 feet east of the site,
- one gradient-control well is installed at well W13 and the ground fluid removed by its operation is managed as described in Section C3.2,
- combined discharge of all wells is approximately 50 to 100 gallons per minute,
- ten observation wells are monitored quarterly,

- collected ground water is discharged to the sanitary sewer as described in Appendix F, and
- the system will be operated for at least 100 years.

Table C4-1 presents capital and operating costs for the system.

C4.5 Advantages and Disadvantages

The following advantages and disadvantages are considered with respect to the gradient-control well system:

Advantages

- The technology for ground-water containment by gradient-control wells is well established,
- the system is highly flexible and can be adjusted as needed with apropriate monitoring,
- disruption of surface activities is minimal, and
- the system limits uncertainty concerning role of multi-aquifer wells and bedrock valley contaminant transport pathway.

Disadvantages

- Requires long-term operation and time frame is undefined (probably hundreds of years),
- collected ground water will require treatment and disposal,
- leakage to the St. Peter may occur even with a gradient control well system in place,
- collection, treatment and disposal costs are significant, and
- additional study is needed to develop final design.

TABLE C4-1
COSTS FOR GRADIENT-CONTROL WELL SYSTEM

	Capital $	O&M $	Years
	Operation and Maintenance		
Data Compilation and Design	$20,000		
2 Production Wells	20,000		
Ground Fluid Removal and Well (100,000 gallons)	85,000		
Sewer Hookup	30,000		
Sewer Discharge Fee (100 gpm)		37,000	100
Monitoring & Reporting ($200/sample)		4,000	100
Operation and Maintenance			
1 Technician ($24/hr)		5,000	100
1 Supervisor ($52/hr)		3,000	100
Energy ($0.06/kw-hr)		1,000	100
Materials (10% construction cost/year)		3,000	100
Real Estate Cost	20,000		
TOTAL	185,000	53,000	100

C5. IN-SITU TREATMENT

Several in-situ treatment techniques were presented and discussed in Section C2. The preliminary screening indicates that there is limited experience with the use of all in-situ treatment techniques. The only in-situ treatment techniques which have been used for reducing hydrocarbon contamination of soils and/or ground water are techniques which promote microbial degradation of select hydrocarbon compounds.

Several terms have been used to describe these techniques. The terms "land treatment" or "land farming" are used to describe the microbial degradation of hydrocarbons in surface soil (generally to a depth of 6 to 12 inches). The term "bioreclamation" is used to describe the in-situ microbial degradation of hydrocarbons in saturated soil and ground water.

Both "land treatment" and "bioreclamation" involve developing a suitable environment for certain aerobic microorganism species which can preferentially utilize specific hydrocarbon compounds as growth substrates. The growth of these microorganisms can be increased several thousandfold by aeration of, and nutrient applications to, the contaminated media. Removal of the hydrocarbon compounds is directly related to the growth rate of the microorganisms.

The petroleum industry has used land treatment to biodegrade select oily wastes for over 25 years. There are approximately 100 full scale and pilot scale facilities of this nature in the U.S. presently (Brown, 1981).

Bioreclamation is a relatively new innovation and is primarily in the research and development stage (EPA 1982). The technique has been applied on a limited scale for the clean up of saturated soils which have been contaminated with fuel oil or gasoline products. These materials tend to "float" on top of ground water so that the bulk of the contamination is restricted to a shallow "band" on top of the ground water. The major volume of the product can therefore be skimmed off the ground water with recovery wells. After the skimming operation, a "band" of contaminated soil remains which can act as a source of continuing ground-water contamination. Several studies have shown that microbial activity can be stimulated in these narrow "bands" and the hydrocarbons adsorbed to the soil can be reduced to acceptable levels (EPA 1982). In addition, hydrocarbons in the ground water which were not removed in the skimming operation may also be removed by the hydrocarbon-utilizing microorganisms.

C5.1 Application

Land treatment is not a viable technology for the site due to the depth of the hydrocarbon contamination. The technique is only applicable for degrading hydrocarbons which are present in unsaturated surface soil. The extensive experience with this technique, however, can provide a perspective on: (a) the feasibility of promoting microbial degradation of the specific hydrocarbon compounds present at the site and in the concentrations observed at the site, and (b) the time frame to achieve removal of the hydrocarbons.

Bioreclamation is a potentially applicable remedial action for the contamination present in the:

- bog deposits
- Middle Drift deposits
- ground fluid (well W13)

Hydrocarbons present in the Middle Drift ground water may be removed also, but the primary objective of bioreclamation is to reduce the hydrocarbons adsorbed to the solid particles by microbial attachment on the solid phase. The ground fluid will be directly affected by the microbial attachment due to the high concentrations of hydrocarbons present in the ground fluid.

The application of the bioreclamation technique and design parameters are extremely site and constituent specific. The method relies on induced or natural hydraulic gradients to distribute the injected nutrients and oxygen. The aquifer characteristics and the depth and extent of contamination will therefore significantly affect the geometry and number of injection and production wells. Figure C5-1 presents a schematic of a bioreclamation system.

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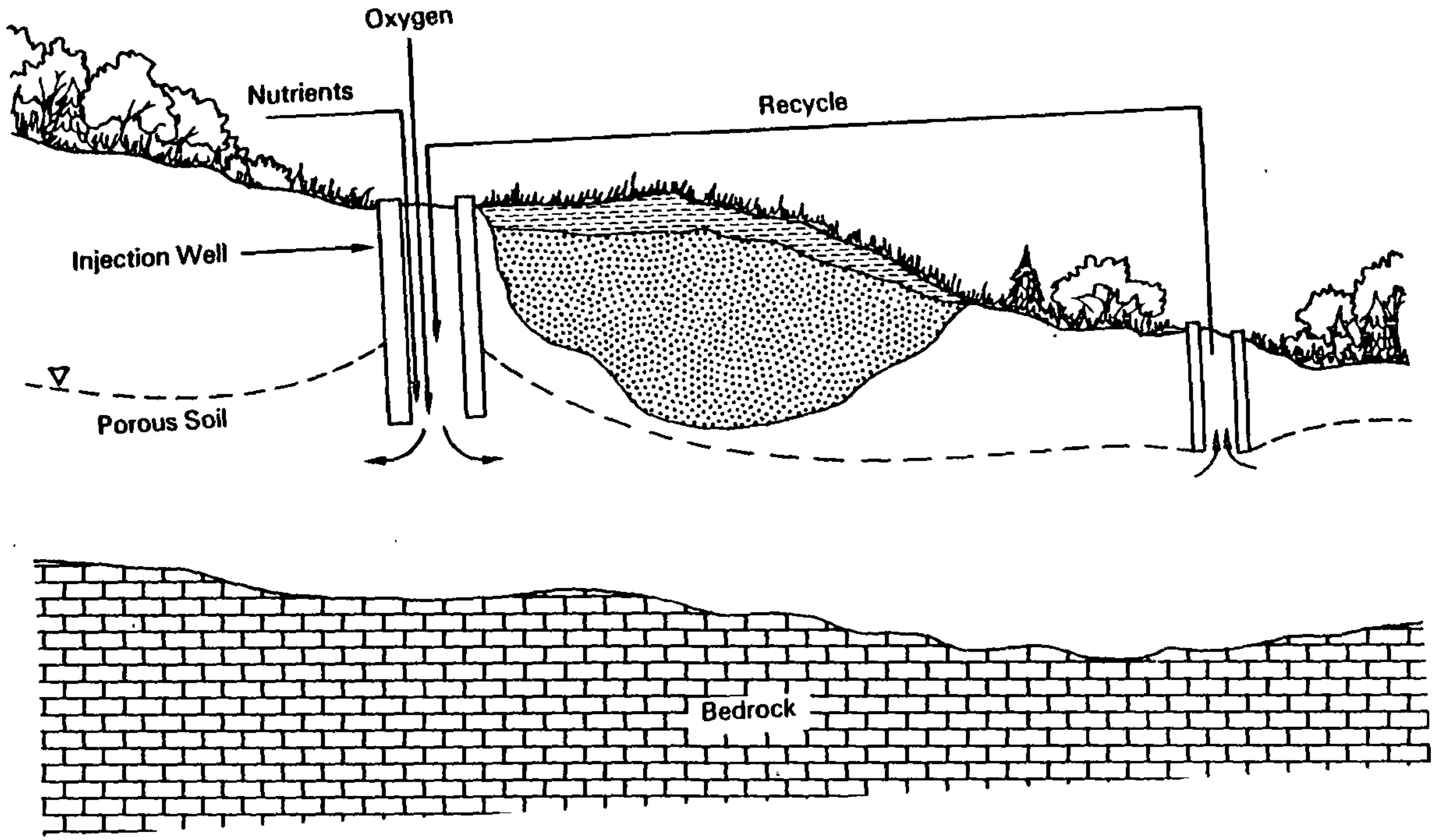

Figure C5-1 Treatment of the Contaminated Groundwater with the Bioreclamation Technique

The types and concentrations of hydrocarbons present will significantly affect the nutrient requirements as well as the time frame and effectiveness of the process. As noted earlier, the process has only been applied on a limited scale to media affected by gasoline contamination of ground water and, in one case, fuel-oil contamination of ground water. The technique has never been used to remove hydrocarbons from media contaminated by PAH and related compounds. Limited laboratory testing has been performed with coal tar derivatives and the data indicate that microorganisms can use polynuclear aromatic hydrocarbons associated with these wastes as a growth substrate (Jamison 1982). There are no data available on the efficiency of removal or the concentrations evaluated, however.

Limited laboratory studies with benzo-a-pyrene suggest that 90% removal of this compound could be achieved in one year through microbial degradation (Shabad 1979). Data from three petroleum industry land-treatment facilities indicate that PAH at land-treatment sites are strongly resistant to microbial biodegradation at concentrations ranging from 25 to 698 ppm (Weldon 1980). An approximate 50% reduction of PAH compounds was observed in these studies over a 3-year period. It has been suggested that the biodegradation of PAH compounds is dependent upon a continued presence of more easily degradable hydrocarbons to support the cometabolic biodegradation of the PAH compounds (Dibble and Bartha 1979). In summary, the limited available experience suggests that PAH compounds can be biodegraded under suitable environmental conditions at concentrations similar to those found in the site soil. The biodegradation of PAH compounds, however, will not be complete and will involve extended time periods. For purposes of economic evaluation it is assumed that the operation of the system would be on the order of 5 years.

C5.2 Design and Operating Considerations

The design components of a bioreclamation scheme include:

- injection wells (for aeration and nutrient injection),
- air compressor and piping,

- production wells and pumps,
- nutrient tanks, and
- nutrients.

The number and spacing of injection and production wells is highly dependent on the hydrogeologic characteristics of the Drift and bog deposits. Because the bog deposits have significantly different hydraulic characteristics than the Middle Drift deposits, the two deposits will probably require separate systems of injection wells and possibly production wells. Nutrient requirements for both systems can be estimated on the basis of total hydrocarbons present in the two deposits and microbial nutrient requirements.

Limitations associated with obtaining adequate oxygen diffusion through heterogenous porous media significantly affect the spacing of injection wells. Experience indicates that the radius of influence for an injection well is approximately 25 to 50 feet in stratified sand and gravel deposits (Jamison 1982). Final design of the bioreclamation system would require pilot scale data to determine the geometry of the wells and the nutrient and aeration requirements.

Bog Deposits

Site characterization data indicates that the bog deposits extend throughout most of the site and vary in thickness from 0 to 33 feet. The bog deposits are underlain by a laterally persistant clayey silt and sand confining layer 1 to 2 feet thick and are covered by artificial fill throughout a large portion of the site. The extent of the contaminated bog deposits and cross sections of the bog deposits are presented in Appendix B.

Limited data are available concerning the hydraulic characteristics of the bog deposits. The stratigraphy of the site and water balances developed for the bog deposits indicate that there is limited movement of ground water in a lateral and vertical direction.

Injection wells therefore may need to be located in the bog deposits to obtain adequate dispersion of injected nutrients and air. Injection wells located in the Middle Drift would not significantly affect microbial activity in the bog deposits. Approximately

100 injection wells should be adequate to disperse air and nutrients throughout a 5 to 10 acre area in the bog deposits.

The presence of a clayey confining bed suggests that production wells located in the Middle Drift will be of limited effectiveness in inducing lateral flow in the bog. Locating production wells in the bog deposits is not considered practical due to the poor water transmitting properties of organic soils. Experience related to the drainage of organic soils indicates that mechanical problems related to clogging of well screens generally occurs when production wells are screened in organic deposits.

Several interceptor drains located in the bog deposits 10 to 15 feet below the ground surface may be adequate to induce lateral flow in the bog deposits. These drains would require a pumping station and should be placed in a layer of sand and gravel and screened with filter fabric to reduce sedimentation. Drain spacings of 300 feet and drain sizes of 8 inches should maintain adequate lateral flow conditions for aeration and nutrient dispersion in the bog deposits. Uneven settling of the drains may occur, due to the poor bearing capacity of organic soils. This problem has been widely discussed and it is generally suggested that organic soils be allowed to "ripen" or settle prior to installation of a final subsurface drainage system (Schwabb et al. 1968).

The water removed by the pumping system can either be disposed of or reinjected. Previous applications of bioreclamation have reinjected the recovered ground water. Reinjection of the recovered ground water could be accomplished by applying the ground water to a leach field located above the water table and allowing the collected ground water to percolate through.

Total hydrocarbons present in the bog and Middle Drift deposits were estimated to evaluate nutrient requirements for a bioreclamation system. Background hydrocarbons are included in this analysis and microbial degradation would not differentiate between background concentrations and those contributed by previous waste-management practices. Table C5-1 presents estimates of the total benzene-extractable hydrocarbons contained in the soil formations underlying areas 1, 2, and 3 of Figure C3-1. The method combines approximate

TABLE C5-1

NUTRIENT REQUIREMENTS FOR A BIORECLAMATION SYSTEM BASED ON AN ANALYSIS OF TOTAL SOIL BENZENE EXTRACTABLE HYDROCARBONS*

Area	Soil Material	Approx. Surface Area (Acre)	Estimated Depth of Material (feet)	Median Hydrocarbon Conclusion (milligrams per kilogram)	Range of Hydrocarbon Conclusion (milligrams per kilogram)	Tons of Hydrocarbons	Nutrients** Required (Tons)
1	Fill	29.0	4.0	9,300	44,400 - 540	2,718	
	Bog	17.4	5.0	15,200	188,000 - 9,890	2,269	
	Drift	29.0	18.0	270	19,500 - 65	267	
2	Bog	6.6	14.0	21,800	140,000 - 1,960	3,049	
	Drift	6.6	19.0	1,875	17,800 - 85	457	
3	Bog	6.6	22.0	10,250	307,000 - 345	2,246	
	Organic Silt	6.6	3.0	2,300	12,300 - 555	108	
	Drift	6.6	26.0	173	10,200 - 65	56.5	
TOTAL	Bog Deposits (inluding fill and organic silt)					10,390	935
	Middle Drift Deposits					780	70

*Estimates include natural background hydrocarbons present

**Nutrients required to stimulate microbial degradation are assumed to be nitrogen (6% of total hydrocarbons present) and phosphorus (3% of total hydrocarbons present) areal boundaries of contamination with the corresponding stratigraphy and soil analytical data to obtain hydrocarbon quantities for the soils in each area. As these estimates are dependent upon subsurface data, assumptions were required between known data points relative to the continuity of stratigraphy and hydrocarbon concentrations.

After delineating the areal distribution for each area, the depth of contamination for the entire soil formation as a unit was determined using the cross sections shown in Figure B5-4 and Figure B5-5 (Appendix B). Basically, the depth of contamination throughout each section extends to the depth where benzene-extractable hydrocarbon concentrations are above background conditions. An analysis was then performed for each area by soil type. Average depths for each soil formation were developed from the population of measurements obtained at the borings throughout each cross section. Subsequently, the median of all available total benzene extractable hydrocarbon concentrations (Figure B5-2 and Figure B5-3), for the respective formations, was assigned to each soil type. At this stage, multiplication of the thickness of each soil type, the corresponding area and hydrocarbon concentration, and an approximate soil unit weight provides hydrocarbon quantities of the soils in the three areas of consideration.

Nutrient requirements for a bog deposit bioreclamation system were estimated based on the following assumptions:

- one gram of hydrocarbons is converted to one gram of microbial mass;
- approximately 10,390 tons of hydrocarbons are in the bog deposits and fill;
- microbial mass is 6% nitrogen and 3% phosphorous; and
- approximately 5 years would be required to convert the hydrocarbons in the bog deposits to microbial mass.

Based on the above assumptions, approximately 125 tons of nitrogen and 62 tons of phosphorous per year would be necessary over a 5-year period. It should be noted that these estimates are only for preliminary cost purposes. Pilot scale studies are necessary to develop accurate estimates of nutrient requirements. These estimates would be adjusted under full scale operating conditions as necessary.

A significant aspect of implementing bioreclamation in the bog deposits is the organic nature of the deposits. Microbial activity will not differentiate between "background" organics and those contributed to the site by past waste disposal. Bioreclamation has never been applied to organic soil and it is not clear what impact background organics will have on the system. Microorganisms have been noted to preferentially utilize less complex organics at land treatment sites, thereby increasing the length of time required for removing more complex hydrocarbon compounds such as PAH's (Weldon 1980). Implementing bioreclamation in the bog deposits may, therefore, (a) increase the nutrient requirements estimated on the basis of total hydrocarbons, and (b) lengthen the total time frame of the project. It is conceivable that most of the organic material present in the bog deposits would have to be converted to microbial mass in order to remove most of the PAH present in the bog deposits.

Middle Drift

A bioreclamation system for the Middle Drift would utilize a gradient control system similar to the one described in Section C5 to disperse the injected air and nutrients. Approximately 100 injection wells would be required to cover an area of 5 to 10 acres. The injection wells would be placed at depths varying from 20 to 60 feet to correspond to the distribution of contamination observed in the drift soil (see Appendix B). Pilot scale testing would be necessary to determine the depth and spacing of the wells.

Nutrient requirements would be approximately 9 tons of nitrogen and 5 tons of phosphorous/year for a 5-year period assuming 780 tons of hydrocarbons present in the Middle Drift.

Pilot Scale Testing

As indicated throughout this section, pilot scale testing is necessary to develop the design criteria for a bioreclamation system. Pilot scale testing would include a limited laboratory study to determine microbial nutrient requirements. A series of injection and production wells would be installed in the Middle Drift and bog deposits to evaluate the efficiency of substrate utilization and air and nutrient dispersion in the deposits. The study would take 6 months to a year to complete and would cover approximately 0.5 acres in the vicinity of well W13.

C5.3 Reliability and Effectiveness

Considerable uncertainties are associated with implementing bioreclamation. The technique has been implemented on a very limited scale and there are few individuals with expertise in this area. The technique has never been applied to PAH and related compounds, organic soil or contamination which is as spatially complex as that found at the site (EPA 1982).

Data from land-treatment systems suggest that significant hydrocarbon concentrations may remain in the bog and Drift deposits even after long-term operation of a bioreclamation system. The effect of residual hydrocarbon concentrations on Middle Drift ground-water quality cannot be quantified due to uncertainties associated with (a) sorption/desorption mechanisms and (b) the efficiency of hydrocarbon removal by bioreclamation.

C5.4 Costs

Costs for a bioreclamation system were developed on the basis of the design considerations presented for the Middle Drift deposits and the bog deposits. The final design of a system would rely on pilot scale results and may vary considerably from the assumptions presented in this section. Due to the high degree of uncertainty associated with the final design and the limited cost data-base for bioreclamation systems, costs are assumed to be accurate within ±50%.

Table C5-2 presents costs for the pilot scale study, the bog deposits and the Drift deposits. All pilot scale costs are assumed to be capital costs due to the short time period involved (0.5 to 1 year).

TABLE C5-2

COSTS FOR BIORECLAMATION

PILOT SCALE

Component	Capital	Operation and Maintenance	Time (Yrs)
Laboratory study	$ 10,000		
1 Production well	5,000		
10 Injection wells	3,000		
2 Interceptor ditch drains	6,000		
Equipment rental	4,000		
Nutrients	1,000		
Sampling and analysis	60,000		
Maintenance	30,000		
Supervision and reporting	20,000		
Total	$139,000		

BOG

Component	Capital	Operation and Maintenance	Time (Yrs)
Interceptor ditch drains and pumping station	$ 20,000		
100 Injection wells	20,000		
Air compressor and piping (200 cubic feet per minute)	20,000		
Nutrient tanks	3,000		
Nutrients		$ 70,000	5
Monitoring and maintenance		60,000	5
Sampling and analysis		120,000	5
Supervision and reporting		30,000	
Total	$ 63,000	$280,000	

TABLE C5-2 (Continued)

DRIFT

Component	Capital	Annual Operation and Maintenance	Time (Yrs)
2 Production wells	$ 10,000		
100 Injection wells	20,000		
Air compressor and piping (200 cubic feet per minute)	20,000		
Nutrient tanks	3,000		
Nutrients	6,000		
Monitoring and maintenance		30,000	5
Sampling and analysis		82,000	5
Supervision and reporting		30,000	5
Total	$ 53,000	$148,000	

C5.5 Advantages and Disadvantages

The principal advantages associated with bioreclamation are:

- the process may significantly reduce contamination associated with the Middle Drift and bog deposits and the ground fluid;
- the implementation time frame is probably short (5 to 10 years);
- the process will also reduce contamination in the Middle Drift ground water; and
- the process can be implemented with minimal disruption of surface activities.

Principal disadvantages include:

- the technology is essentially unproven for the type of contamination present at the site;
- a significant research and development effort would be required to develop design parameters for a bioreclamation program;
- limited technical expertise is available for this technology;
- even after conducting detailed pilot scale testing, considerable questions would still exist regarding the efficiency of hydrocarbon removal, the time required for satisfactory degradation of contaminants and the impact of the residual hydrocarbon contamination on the Middle Drift ground-water quality; and
- bioreclamation has never been used on organic soils and it is uncertain what effect background organics would have on microbial degradation processes. Dispersion of the injected nutrients and air may be difficult to accomplish in the peat and organic soils due to their poor permeability.

The high level of uncertainty associated with the design, costs, effectiveness, practicability and operating time of bioreclamation do not justify expending resources on this technology at the site. The concept is attractive due to its relative "simplicity", however, considerable research and development must be conducted on this technology in order to consider it a viable alternative.

C6. MONITORING AND SITE-USE CONTROLS

C6.1 Application

Monitoring and site-use controls are remedial actions which directly address the objective of allowing for the safe future use of the site. In addition, monitoring is a potential option for addressing the issue of contaminant transport. Data presented in Appendix B and modeling results presented in Appendix E suggest that contaminant transport from the site should not affect the selection, operation or cost effectiveness of off-site controls or their reliability and effectiveness in meeting the objectives of: (a) providing St. Louis Park with a safe and adequate supply of potable water, and (b) allowing for the present and future use of ground-water resources in the St. Louis Park area as sources of public drinking water supply.

Although the Drift-Platteville is hydraulically connected to lower bedrock units, it is unlikely that this connection will affect the type or extent of off-site controls recommended for municipal production wells due to the (a) long hydraulic travel times involved, (b) effects of degradation, dispersion and retardation, and (c) the location of the wells relative to hydraulic connections and aquifer gradients.

A comprehensive monitoring and site-use control plan is presented in this section to address the issues concerning the safe future use of the site as well as uncertainties concerning the effect of contaminant transport on the reliability and effectiveness of off-site controls.

C6.2 Design and Implementation Considerations

Site-Use Considerations

The site is currently used for a variety of residential and commercial purposes without any apparent adverse impact. There is no contamination exposed at the surface. Moreover, there are no signs of vegetation stress or nuisance conditions such as odors. The most significant aspect of site-use which the site contamination has affected is the extension of Louisiana Avenue through the bog area. It is believed that the political complexities resulting from the discovery of the site contamination may have stalemated any further action on this project due to the uncertainties concerning the eventual outcome of remedial actions at the site area.

Plans for road construction and other development can proceed if future developers are aware of the site characteristics and if they take appropriate engineering and design precautions. Because of the naturally low bearing capacities of the organic soil associated with the bog, any construction in or through that area might require the use of piles or bridges. The removal of this poor quality soil and replacement with fill of a higher bearing capacity would probably be impractical even if it were uncontaminated because of the depth to which it occurs. Surcharging, dewatering or other methods commonly used to consolidate similar organic soil should be undertaken with due regard to the nature of the ground water that would be driven or pumped out of the sediments. Therefore, any plans to construct the highway interchange in the bog should proceed carefully considering the contaminated nature of the soil in this area. In addition, a worker safety education program should be developed to educate workers concerning appropriate procedures and precautions for the excavation and handling of contaminated soil.

Contaminant Transport Considerations

The Drift-Platteville aquifer has been affected by the site contamination, although the precise extent of Drift-Platteville contamination is unknown. A survey of private wells, currently underway, indicates that there are over 1,000 private wells in the St. Louis Park area. Since the entire area is connected to a municipal water supply, many of these wells have been abandoned although some are still serviceable and may be used for non-potable purposes. The presence of these wells suggests that contaminant transport in this aquifer is a potential concern if the contamination which may be present in private wells is not a matter of public record. Futhermore, a municipal well (well SLP3) is open to the Platteville and St. Peter. This well is not contaminated; however, modeling indicates that the well could possibly be affected in the future should contaminant transport occur in the St. Peter aquifer.

Appendix B describes the extent of contamination in the Drift-Platteville. Limited data indicate that the Drift-Platteville contamination extends 1,400 feet due east from the site and that the area of contamination is as wide as the contaminated soil area (that is, the southern half of the plant site and the bog). Wells open to the Drift-Platteville and St. Peter are potential conduits of contaminant movement from the Drift-Platteville to the St. Peter. This potential exists as a result of the vertical head difference; however, flow between the two aquifers has not been detected in any multi-aquifer well open to these two aquifers.

The extent of contamination in the Drift-Platteville aquifer should be clearly identified and a ban on the use of this aquifer as a potable water supply or for gardening or food processing uses (see Appendix H) be established in the area of known contamination. Water quality downgradient from this area should be monitored to identify whether and at what rate contaminants are moving downgradient. Development of specific ground-water use control recommendations should be conducted after evaluating:

- all existing data on ground-water quality in the Drift-Platteville,
- any available unpublished USGS data on hydrology and water quality parameters in the Drift-Platteville, and
- results of the well survey data.

The USGS data should be used to identify the extent of contamination in this aquifer more accurately. The well survey data should then be used to (a) identify wells within the contaminated area which should be abandoned or closed and (b) identify an appropriate area in which the use of these aquifers for private potable purposes is not recommended.

A comprehensive ground-water monitoring plan should be developed based on the extent of ground-water contamination. The monitoring plan would include the sampling of wells within the contamination area and downgradient of this area. In addition, two multi-level monitoring wells should be installed in the Drift-Platteville and St. Peter in a straight line between the north east corner of the site and well SLP3.

The comprehensive monitoring plan will identify appropriate monitoring frequencies and parameters, and will provide a statistical basis for determining if contaminant transport is occurring. A contingency plan will be incorporated in the comprehensive monitoring plan to identify appropriate responses to statistically significant changes in water-quality parameter concentrations. These contingency responses could include decreasing or increasing the area over which a drinking water use ban is extended and recommendations concerning the future need for abandoning or treating well SLP3.

For cost estimating purposes, it is assumed that the comprehensive monitoring plan would include the installation, sampling and analysis of 10 new multi-level monitoring wells and the sampling and analysis of 10 existing wells. The new monitoring wells include those used to monitor ground-water quality between the site and SLP3. The wells would be analyzed for parameters such as phenolics, total organic carbon (TOC), sodium, total dissolved solids (TDS) and select PAH. An annual to biannual sampling frequency is assumed.

<u>Summary</u>

Specific actions which should be taken to develop appropriate monitoring and site-use controls include:

- prepare a worker education program for the safe development of the bog area as a highway interchange;
- evaluate and integrate unpublished USGS data from the well survey to identify potentially impacted areas;
- develop specific recommendations concerning abandonment or closure of private wells and identify an area over which private drinking water use should be banned; and

- develop and implement a comprehensive ground-water monitoring and contingency plan for the impacted area.

C6.3 Effectiveness

The monitoring and site-use control program is highly effective for ensuring the safe future use of the site. The program also ensures the safe reasonable use of the Drift-Platteville and St. Peter. The program does not minimize contaminant transport, but provides necessary data to determine what, if any, contaminant transport is occurring in the Drift-Platteville and St. Peter and its potential effects.

C6.4 Costs

Costs for the monitoring and site-use control program are presented in Table C6-1. Costs for the site-use control plan assume that a site use report will be prepared identifying the location and extent of contamination and appropriate engineering and design precautions that should be observed due to the subsurface contamination during construction.

The comprehensive monitoring plan includes the review of the well survey results and unpublished USGS data, development of a methodology for evaluating the data and a contingency plan, the installation of up to 10 new multi-level wells and the semi-annual sampling and analysis of up to 20 wells.

C6.5 Advantages and Disadvantages

The following advantages are associated with the monitoring and site-use control plan:

- it is the only site remedial action which addresses the safe use of the site and the contaminated ground water in the vicinity of the site;
- it can be easily implemented and is highly effective in providing for the safe future use of the site;

TABLE C6-1

MONITORING AND SITE USE CONTROL COSTS

	Capital Cost ($)	Annual Costs ($/Year)
Studies		
Develop Worker Education Program	20,000-40,000	
Develop Ground Water Monitoring Plan	20,000-40,000	
Develop Recommendations Concerning Drift-Platteville Use	20,000-40,000	
Monitoring		
Install 10 New Multi Level Wells	20,000-50,000	
Analyze 20 Multi Level Wells*		7,000-14,000
Sampling and Data Reporting		6,000-10,000
	80,000-170,000	13,000-24,000

*Costs are for analyzing 60 samples at $200/sample annually or biannually and a 20% contingency for replicates and split samples.

- continued long term monitoring can address uncertainties concerning whether and at what rate contaminant transport is occurring and the effect of contaminant transport on the reliability and effectiveness of off-site controls.

The principal disadvantage is that the site contamination remains in place and is not contained.

C7. SUMMARY AND CONCLUSIONS

C7.1 Objectives

As noted in Section C1, the objectives of any remedial action implemented in St. Louis Park are to allow for the appropriate present and future uses of the aquifers and the site area and provide a safe drinking-water supply for St. Louis Park. Site remedial actions are evaluated in this Appendix with respect to their cost effectiveness in reducing contaminant transport from the site and allowing for the safe, reasonable and beneficial uses of the site in the foreseeable future. The objective of reducing contaminant transport was established for site remedial actions as an indirect means of assisting off-site controls in achieving the principal objectives of (a) providing a safe drinking-water supply for St. Louis Park and (b) allowing for appropriate present and future uses of the aquifers.

The objective of remedial action is not to minimize impacts on the Drift-Platteville aquifer. This aquifer is not considered a significant present or future public drinking-water supply source. The Drift-Platteville aquifer, however, is a potential conduit of contaminants from the site to the deeper bedrock aquifers. Available monitoring data and modeling results suggest that this pathway may not be a significant source of contamination to the deeper bedrock aquifers due to degradation, dispersion, and retardation effects. Implementation of "source" controls, therefore, is not expected to provide a significant benefit with respect to allowing for present and reasonably foreseeable future uses of ground-water resources in the St. Louis Park area or providing St. Louis Park with a safe drinking-water supply.

A significant monitoring burden, however, is incurred to ensure that the above conclusion is valid. The cost of monitoring and additional studies must therefore be compared to the cost of source controls which can limit the monitoring study burden.

Existing site conditions do not result in present or projected adverse impacts due to off-site migration of site contaminants via air and surface water. The vast majority of the site has already been covered, supports normal types of vegetation and is being used for community and private purposes. No remedial action is necessary to allow for safe, present, reasonable and beneficial use of the site. Plans for road construction and other development can proceed if future developers are aware of the site characteristics.

C7.2 Effectiveness and Feasibility of Remedial Actions

Section C2 presents the range of potential remedial actions considered to remove, contain, or treat the site contamination. The evaluation indicates that excavation, pumping of the ground fluid at well W13, bioreclamation, gradient-control wells and monitoring are technically feasible options which could potentially meet the study objectives. These options were evaluated in detail to evaluate the cost, effectiveness and reliability of the individual options.

Each option varies in terms of effectiveness (as defined by reducing the potential transport of contaminants to the deeper bedrock aquifers and allowing for present and future uses of the site), and technical feasibility (as defined by the previous application of the technology to similar contamination problems).

Excavation of contaminated soil is technically feasible although the effectiveness is poor to moderate for the site conditions. Significant amounts of contamination would still remain in the Middle Drift deposits due to limitations associated with the depth of excavation and the presence of buildings and roads. This contamination could still be transported to the deeper bedrock aquifers. Removing the contamination, however, would ensure the safe future use of the site although excavation would result in significant disruption of present site use.

Removal of the ground fluid is technically feasible and may reduce contaminant transport. This option, however, should be implemented in conjunction with a gradient-control well system due to the potential desorption of PAH and related compounds from drift deposits which were in contact with the ground fluid prior to its removal. Removal of the ground fluid would not result in a significant benefit in regard to the future use of the site.

The technical feasibility of bioreclamation is highly questionable. Limited experience is available with this technique and it has never been used at sites contaminated by PAH and related compounds or implemented in organic soils. Furthermore, available data concerning microbial degradation of oily wastes by land treatment indicates that PAH removal would not be complete. The effectiveness of this technique in reducing contaminant transport is considered poor due to incomplete biodegradation and implementation problems.

Gradient-control wells are technically feasible. The effectiveness of a gradient-control well system with respect to reducing contaminant transport is considered moderate due to leakage effect. The gradient-control well system does not result in any significant benefits related to ensuring the safe future use of the site.

Monitoring is not effective in reducing contaminant transport to the bedrock aquifers. Monitoring, however, is highly effective in ensuring the safe future use of the site.

C7.3 Costs

The costs of the remedial actions evaluated vary significantly in terms of capital and operating costs. All costs were converted to present worth costs as discussed in Appendix F to account for the variation in operating times. Table C7-1 summarizes the present worth costs for the remedial actions evaluated. A range of costs is presented for the gradient-control well system which reflects the difference in present worth costs for pumping 50 gallons per minute or 100 gallons per minute for 100 years. The range of costs presented for monitoring reflect the difference in sampling and analyzing 20 monitoring points for a period of 20 years to 100 years. The range of bioreclamation costs reflect the difference in operating the system for 1 to 10 years.

C7.4 Evaluation of Remedial Actions

The most cost effective option with respect to allowing for the safe future use of the site is the monitoring option. Only one other

TABLE C7-1

COSTS OF REMEDIAL ACTIONS

Remedial Action	Present Worth Costs (Dollars)
Excavation	205,000,000 to 333,000,000*
Ground Fluid Recovery	85,0000
Gradient-Control Wells	860,000 to 1,230,000
Bioreclamation	770,000 to 3,560,000
Monitoring/Site Use Controls	340,000 to 650,000

*Partial cost does not include items such as removal and replacement of structures, roadways and utilities, engineering and design, wastewater pumping treatment or dispersal, land owner compensation, and legal fees.

source control option (excavation) was considered applicable to this objective. The costs and socioeconomic factors related to the excavation option, however, clearly outweigh the benefits.

The most cost-effective remedial action with respect to reducing contaminant transport to bedrock aquifers is the implementation of a gradient-control well system. Contaminant transport to bedrock aquifers, however, does not in itself represent a danger to the environment or the public health and welfare. The danger is related to the ultimate use of the potentially affected aquifers. Thus, implementation of a gradient-control system must only be considered if it is cost effective when compared to off-site control options. This analysis is presented in Chapter 7 of the report.

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APPENDIX D
MULTI-AQUIFER WELL CHARACTERISTICS AND REMEDIAL ACTION

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D1. INTRODUCTION

D1.1 Multi-aquifer Wells

A multi-aquifer well is a well which provides a hydraulic connection between more than one hydrogeologic unit. The connection may be by design, as when a well is screened or open to more than one aquifer in order to improve well yield. Inadvertent connections also occur in wells which are ungrouted, in poor condition or improperly constructed. For example, cased but ungrouted wells may permit flow through the annular space between the casing and rock walls of the well bore.

The U.S. Geological Survey (Hult and Schoenberg 1981) has emphasized the potentially significant role that multi-aquifer wells have played in spreading contamination from the surface aquifer to bedrock aquifers in St. Louis Park. In the Twin Cities hydrogeologic basin, the potentiometric elevation is generally higher in shallower aquifers, thus there is a positive head gradient downwards between aquifers. In a multi-aquifer well, the head gradient will tend to produce flow from the upper aquifer to the lower. If the ground water in the upper aquifer about the well is contaminated, the multi-aquifer well flow will convey contaminants to the lower aquifer.

The complete extent and significance of multi-aquifer well flow in the St. Louis Park area is unknown. Geophysical and television logging of thirty multi-aquifer wells performed by the U.S. Geological Survey (USGS) and the Minnesota Department of Health (MDH) has found five wells which have significant interaquifer flow (up to 150 gpm), but flow could not be detected in the remainder (Hult and Schoenberg, 1981). The significance of multi-aquifer wells to the contamination problem is not yet established by field information. While certain multi-aquifer wells are likely to be significant -- most prominently, the on-site wells -- others outside the zone of contamination may be of little or no significance. The significance of multi-aquifer wells in contributing to contamination problems must also be viewed in the context of other remedial actions to be completed. If, for example, ground-water contamination is confined to a certain area by control systems, then multi-aquifer wells beyond that area are not of concern. Therefore, the design of a remedial program for multi-aquifer wells must necessarily be done in the context of a comprehensive remedial action plan. Such a plan is proposed in the main body of this report. This appendix supplies background data and cost estimates for multi-aquifer well remedial actions as background to the main report.

D1.2 Plant Site Multi-aquifer Wells

Two wells on the plant site, W23 and W105, were drilled to a depth of over 900 feet, thus they penetrate the Mt. Simon Sandstone and every above-lying aquifer in the Twin Cities basin. Well W23 was the supply well throughout the life of the RT&CC plant. In 1979 well W23 was found to be plugged at a depth of 595 feet, with water entering the well through holes in the casing at around 215 feet (basal St. Peter) and exiting the well through holes in the casing at 264 feet (Prairie du Chien). A core of the top one foot of the plug revealed a black, viscous, coal-tar-like material. A coating of what appeared to be similar material was seen on the rock surfaces of the Prairie du Chien in the vicinity of the hole at 264 feet (Hult and Schoenberg 1981).

The phrase coal-tar-like material is used to describe the contamination in well W23. This material resembles coal tar in its general appearance, however, there have not been any tests done on this material which identify it as coal tar. The material in W23 may represent sludge bottoms, tank cleanings, weathered coal tar, water gas tar, or any of these materials mixed with each other, creosote, and/or sand. Therefore, for lack of precise knowledge of what went into W23, this material is called coal-tar-like material.

The other plant site multi-aquifer well is W105 which was used as a source of water for the sugar beet refinery from 1898 to 1905 (Larkin 1917a). The well was not used by RT&CC and was abandoned in 1933 (McLellan 1934). The nature of the abandonment and the present condition of the well are not known. Also, it is not known if the well is, or ever has been, a pathway or a source of contamination to bedrock aquifers, although the possibility exists that well W105 could be as important as W23 in this regard.

Section D2 of this appendix provides a description of W105 and W23 based on our current knowledge of these wells. The description is focused on how the wells may act as sources of contamination to bedrock aquifers. Based on this information, section D3 provides suggestions for controlling the contaminant source on an aquifer-by-aquifer basis. The decision to control these sources must be based on the ability of source controls to achieve the overall goals of providing safe drinking water in a reliable and cost-effective manner and allowing for present and future use of the area's ground-water resources. Section D4 discusses off-site multi-aquifer wells and section D5 discusses alternative multi-aquifer well remedial actions.

D2. PLANT SITE MULTI-AQUIFER WELL HISTORIES

The following history of well W23 is based on well logs and RT&CC records, as well as information resulting from site investigations since the plant ceased operations in 1972. The historical information is confusing due to conflicting and incomplete records. For example, a well log exists for a 950 foot deep well owned by Republic Creosoting (RT&CC) which was drilled in 1908. However, as indicated in Appendix A, RT&CC did not locate in St. Louis Park until 1917. In light of possible errors such as the 1908 well log, a chronology of W23 has been compiled using the best available information. The information which is most accurate is that which has been developed during site investigations since the plant closed, when observations and measurements of the well have been carefully recorded.

The W23 history presented in this appendix is based on facts that are for the most part, supported by records from later years, especially the well investigations performed starting in 1979. The brief review of W105 contains only that information for which there is adequate documentation.

D2.1 W23 History During Plant Operations

The well log which is believed to be the correct log for W23 is for a 909 foot deep well which was constructed from December 10, 1917 to May 13, 1918. A copy of this log is shown in Figure D2-1a. Attached to this log is another sheet (Figure D2-1b) which indicates that the driller did some repair work on some shallow wells originally built for the sugar beet refinery and, more importantly, the driller cleaned out the deep well drilled for the sugar beet company by Swenson Well Co. in 1889. The significance of this statement is that the well log for the 909 foot well is for a new well that was drilled in 1917-1918 (W23), and it could not be the log for the older sugar beet well (W105). The driller's remarks about work on the shallow wells and the deep sugar beet well are confirmed by company correspondence discussing these activities, as is the fact that the company had a new well (W23) drilled by McCarthy Well Company at the time (Larkin 1917b, 1917c, 1917d).

Location ______ Town St. Louis Park

Date Started Dec. 10, 1917 Machine No. 3 State Minn.

Date Completed May 13, 1918 Owner Republic Cresoting Co.

File No. ______ Total Depth of Well 909

DIAMETER OF HOLE	20	16	12	10	8⅞	6	4½	TOTAL
Top of Pipe Below Surface			2	80	227			
Bottom of Pipe Below Surface			65	257	373			
No. of Ft. of Pipe in the Hole			63	177	145			
No. of Ft. of Hole Drilled			258	115	536			909

	TEST 1	TEST 2	TEST 3
Depth of the Hole	909	909	909
Depth to Water at Rest	46	46	17
Depth of Water Pumping	63	80	27
Depth of Pump Pipe	63	87	47
Size of Cylinder	8	8	8
Length of Stroke	28	34½	34
Strokes per Minute	25	43	49
Gallons per Minute	150	300	330
Will Well Supply More			Yes
Was Strainer in Hole			
Hours Putting in Pump			
Hours Pumping			
Hours Taking out Pump			
Hours Consumed			

STRAINER

Make			
Diameter			
Total Length			
Number			
No. of Ft. Exposed			
Was Str. Sedged			
Did Sand come thru Str.			
Was Str. Coarse Enough			
Style of Fittings			

FORMATION	THICK-NESS	DEPTH
Limestone & Gravel	60	60
Limestone	35	95
Sand Rock	100	195
Red Shale	15	210
Sand Rock	4	214
Red Shale	6	220
Sand Rock	6	226
Red Shale	3	229
Sand Rock Shaley	29	258
Hard Rock	114	372
Sand Rock	85	457
Shaley Sand Rock	50	507
Shale	138	645
Shale Sand Rock	67	712
Shale	69	781
Shaley Sand Rock	54	835
Sand Rock	72	907
Shale & Sand Rock	2	909

NOTE: The third test was made after the 10" pipe was cut off 80 feet — 6" below the surface.

Figure D2-1a Reproduction of Original Well W23 Driller's Log

8303049

Location ____________________ Town St. Louis Park

Date Started Sept. 29, 1917 Machine No. State Minn.

Date Completed Dec. 8, 1917 Owner Republic Cresoting Co.

File No. ____________________ Total Depth of Well Repair

DIAMETER OF HOLE	20	16	12	10	87'	6	4½	TOTAL
Top of Pipe Below Surface								
Bottom of Pipe Below Surface								
No. of Ft. of Pipe in the Hole								
No. of Ft. of Hole Drilled								

	TEST		
	1	2	3
Depth of the Hole			
Depth to Water at Rest			
Depth of Water Pumping			
Depth of Pump Pipe			
Size of Cylinder			
Length of Stroke			
Strokes per Minute			
Gallons per Minute			
Will Well Supply More			
Was Strainer in Hole			
Hours Putting in Pump			
Hours Pumping			
Hours Taking out Pump			
Hours Consumed			

STRAINER

Make			
Diameter			
Total Length			
Number			
No. of Ft. Exposed			
Was Str. Sedged			
Did Sand come thru Str.			
Was Str. Coarse Enough			
Style of Fittings			

FORMATION	THICK-NESS	DEPTH

NOTE: This work was done on a number of shallow wells approx. 60 feet deep each, which were originally drilled for the old sugar factory about 1898. No success was made with these shallow wells so we pulled the screens and gave them credit for $25.00 per well for the screens. We also cleaned out the deep well drilled by Swenson Well Co., to a depth of 1,000 feet in 1889 and recased it with 12 inch and 8 inch pipe.

Figure D2-1b Reproduction of Original W23 Well Log Repair Work

An inspection of the well W23 log shown in Figure D2-1a reveals important information about sources of water to the well. Three pumping tests were performed at the time the well was completed, with the third test made after the top 80 feet of the ten-inch casing was removed. This meant that the well was an open hole (no casing) from 65 to 80 feet and that water from the Platteville Formation contributed to the well. This fact is supported by the shallower depth to water in the well under both static and pumping conditions as noted for test three on the log. The shallower depth to water made it less expensive to operate the air lift pumping system used on the well. Figure D2-2 is a sketch of well W23 based on the information in the well log. The air lift system was a simple arrangement in which compressed air was introduced through one pipe, forcing water to the surface in another pipe. There are no records of the pumping rate during plant operation; however, based on requirements to operate the plant, the pumping rate was on the order of 50 gallons per minute on a continuous basis (See Appendix A). Figure D2-2 indicates a range of depths for the casing schedule. Lesser numbers are the depths reported on the well log, while the larger numbers are depths determined during recent investigations. The discrepancy is generally about five feet, which may represent a change in ground surface elevation due to fill materials.

During the early life of the plant, W23 discharged to a pond near the refinery building and from there water was pumped throughout the plant (see Appendix A for plant layout). A description of this system was given in a March 27, 1942 memo which stated that the water from the well spilled into a rectangular tank which overflowed into the pond (Mootz 1942). The well was located outside the southeast corner of the refinery building as shown on the plant site drawing in Appendix A (Figure A3-1).

In 1933, new casing was added to the well on the advice of McCarthy, the well driller, and Bradley, the St. Louis Park village engineer. This work was done in response to water quality problems in the first municipal well drilled by St. Louis Park in 1932. While no evidence was offered as to why the municipal well had problems, there

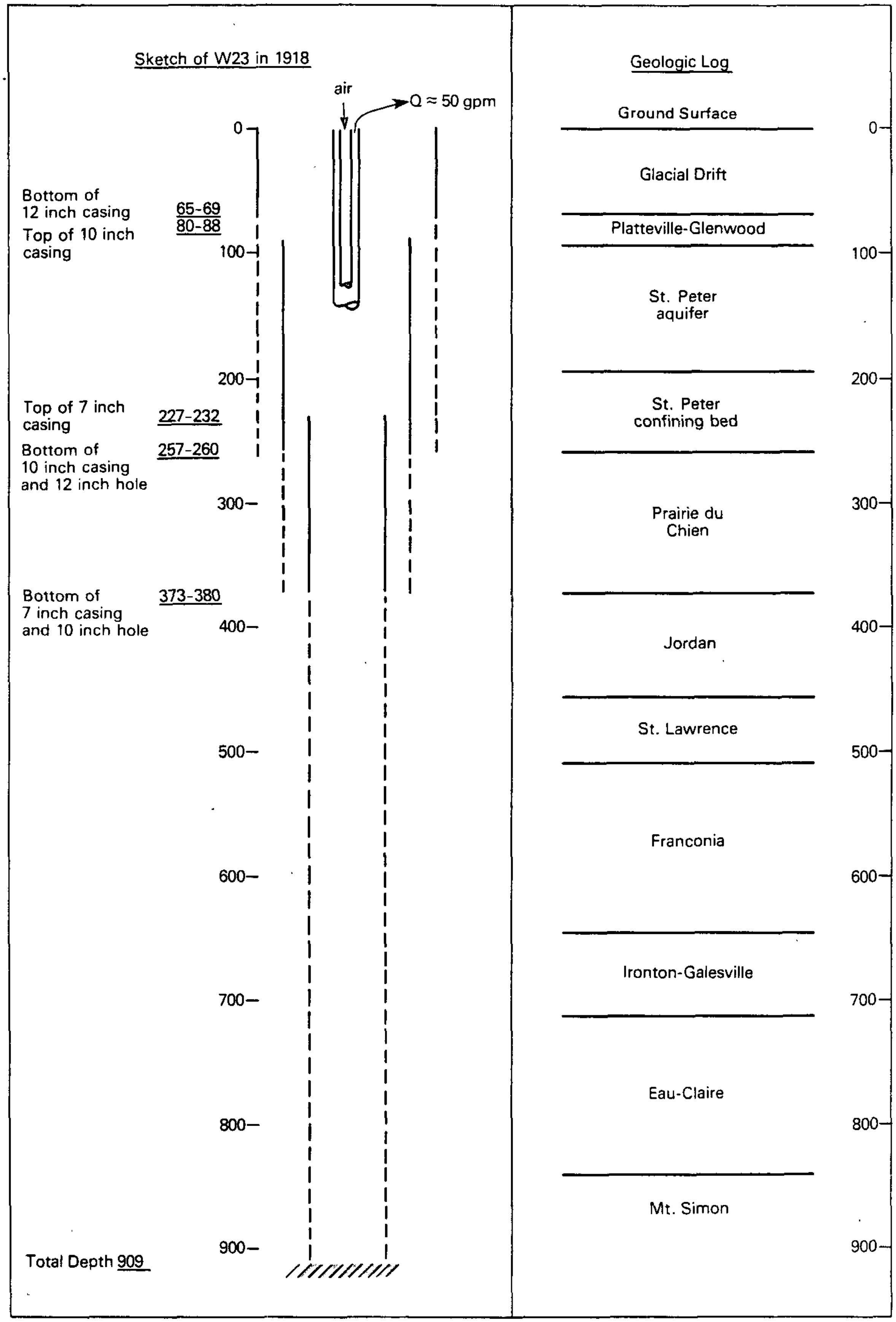

Figure D2-2 Sketch of W23 Based on Original Well Log

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was speculation that the RT&CC plant was responsible (Larkin 1933 and McCarthy 1934). New ten-inch casing was installed in W23 from the ground surface to 80 feet. There is no description of this work in the records, however it is inferred from company correspondence dated May 10, 1933 which includes the statement, "This well has now been cased at the 65 foot level and the casing is continuous now to a depth of 373 feet at which point the casing ends". (Rademacher 1933). The nature of the work was discovered on November 29, 1982 when the top 88 feet of ten-inch casing was removed from the well. An inspection of this casing showed that it was obviously an add-on segment. The discrepancy between 80 and 88 feet is attributed to measurement error, changes in ground surface elevation, and a casing extension of about two feet by the USGS in 1979.

The next change made at W23 was in the mid-1950's, probably 1955. The pumping system was changed from the air lift method to a hydropneumatic system in order to bypass the pond as a reservoir (Holstrom 1955).

Company correspondence in 1958 indicated that after the hydropneumatic system had been in operation for about one year problems developed with the pump because of "tar" deposits on the bearings (Holstrom 1958). Holstrom indicated that the cause of this contamination was unknown. In an effort to remedy this problem, four and one-half inch casing was installed through the Prairie du Chien Formation and eight feet into the Jordan Formation with a seal attached to the lower end (Holstrom 1958). After 1958, there are no more references to the well or pumping system in the company records. The well pump was salvaged by RT&CC when the plant was razed in 1972.

D2.2 W23 History After Plant Closing

Barr Engineering Co. describes their investigation of W23 in their Phase II Report (Barr 1977). Barr removed the well head and pulled a length of pipe out of the well. The pipe had "noticeable coal-tar material" below the water level (about 40 feet below ground surface) which thickened with depth (Barr 1977). Barr Engineering Co. pumped W23 and took water samples after 1, 15, 30 and 100 minutes. The phenolics content of the samples were 0.020, 0.011, 0.008, and

0.005 milligrams per liter, respectively, indicating a decrease in phenolics concentration with time. This relatively rapid decrease is a typical pattern when uncontaminated (or less contaminated) water is available to mix with contaminated water in the well. In the case of W23 this pattern probably resulted because the well is contaminated internally.

In 1979, the USGS made geophysical logs of W23 including natural gamma, caliper and flow logs, and a TV log was made at the same time (Hult & Schoenberg 1981). As part of this work, the casing installed in 1958 was removed. The results of these logs are shown in Figure D2-3. The TV log showed that the well was coated with coal-tar-like material at various locations along the well bore, and this coating was seen in a solution channel in the Prairie du Chien Formation, as viewed through a hole in the casing at about 264 feet. The TV log also showed water entering the well through holes in the casing adjacent to the St. Peter Formation and leaving W23 through holes in the casing adjacent to the Prairie du Chien Formation. This flow of water was measured to be about 150 gallons per minute. The USGS also took a core sample of the bottom of the well from 595 to 596 feet and found mixed sand and "tar" (coal-tar-like material) (Hult and Schoenberg 1981). After this work was done, the City of St. Louis Park had a packer installed in the well at about 250 feet to prevent the flow of water from the St. Peter to the Prairie du Chien (Hult and Schoenberg 1981). After interaquifer flow was shut off in W23, the USGS collected a sample for subsequent PAH analysis by Midwest Research Institute (MRI). The sample represented water from the Prairie du Chien-Jordan aquifer plus any water that may have been drawn from the St. Lawrence and Franconia Formations to a depth of 595 feet. MRI found a total of 328 micrograms per liter of total PAH in that sample (MRI 1981).

It is not known how or when the plug of material got into W23. A careful review of plant records has not revealed any information about the orgin of this material. The rumor related by Barr (1977) and by Hult and Schoenberg (1981) that a railroad tank car overturned and spilled its contents down W23 cannot be supported by documentation. There were no railroad tracks in the vicinity of W23 (see plant layout,

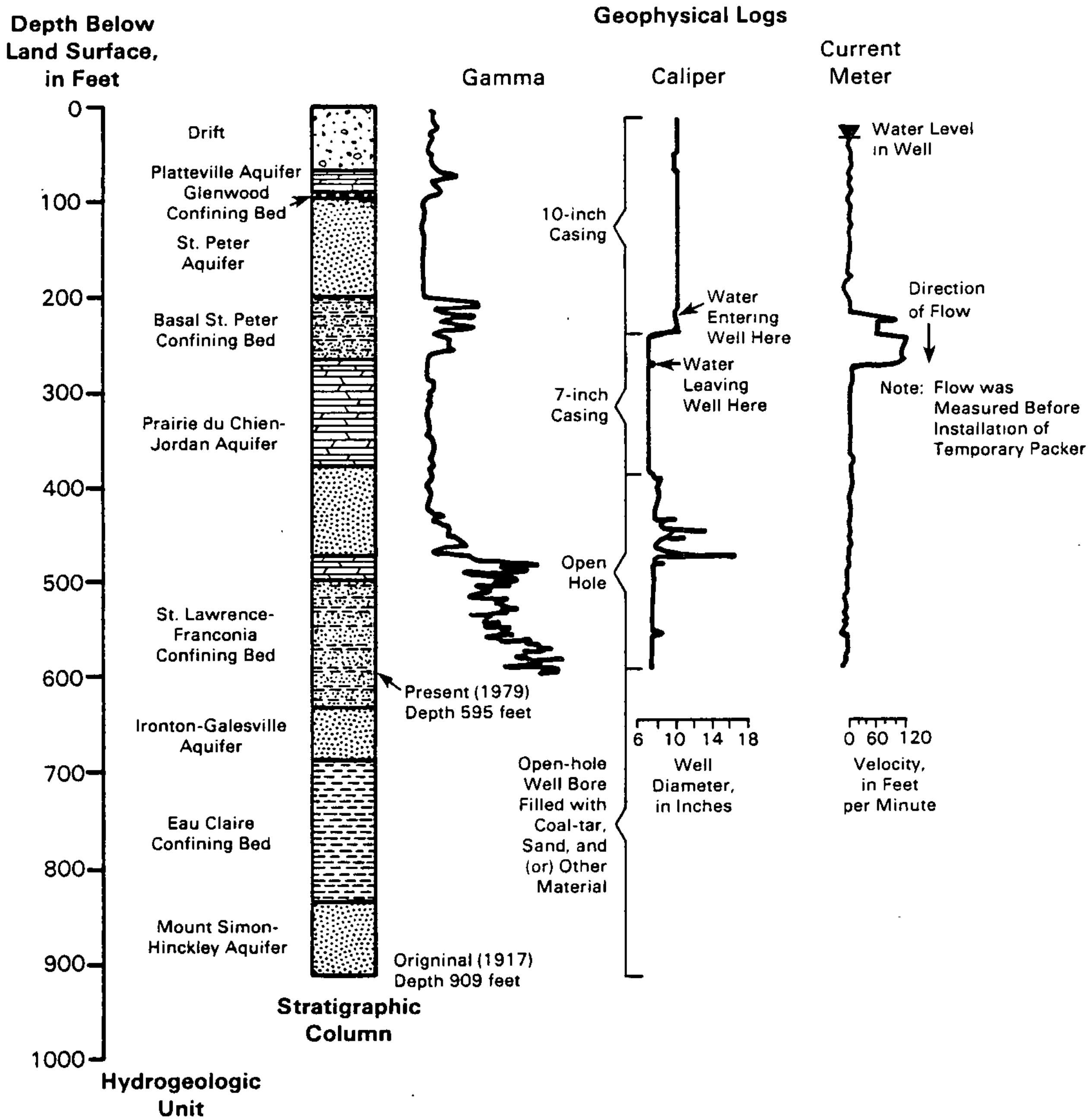

Figure D2-3 Geologic and Geophysical Logs of Well W23 (Reproduced From Hult and Schoenburg, 1981)

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Appendix A), and the viscous incoming raw coal tar typically had to be heated with steam coils before it could be transferred from tank cars (see plant flow sheets, Appendix A).

In addition, nothing that has been found in W23 has indicated the timing of when the coal-tar-like material entered the well. It is hoped that further exploratory work and/or analyses of samples previously collected from W23 will provide answers to this question.

In June 1982, the Minnesota Pollution Control Agency (MPCA) began investigations in W23 of the plug of material beginning at the 595 foot level. The contractors working for the state included E. A. Hickok & Associates and Renner Drilling Company. Also present during this work were representatives of Soil Exploration Company, who kept records of the work and split samples with the MPCA's contractors.

From June 7 to July 29, 1982 the work involved sampling and removing the material in the well bore from 595 feet to 866 feet using cable tool methods. A four-inch liner was installed in W23 from the ground surface to 595 feet. A packer was placed at 250 feet to prevent interaquifer flow, and another packer was placed at the bottom of the four-inch liner to isolate the plug from the rest of the well. The cable tools were used inside the four-inch liner to remove the plug below 595 feet. A log of material found in this interval and a record of a TV log are shown in Figure D2-4. Approximately 290 gallons of material were removed based on the dimensions of the plug in the well, namely, a cylinder 140 feet long and seven inches in diameter. As shown in Figure D2-4, the interval from 740 to 780 feet was void: there was no material of any kind in this zone. In this interval the adjacent Eau Claire Formation is mostly shale. Subsequent TV pictures showed shale plates breaking off the well bore and falling down the hole. It is believed that shale slumping into the well formed a bridge at 740 feet before coal-tar-like material entered the well, and that all the material, throughout the history of the well, remained above the 740 foot level. When the well was being investigated below the 740 foot level some coal-tar-like material followed the driller's tools to the bottom of the hole.

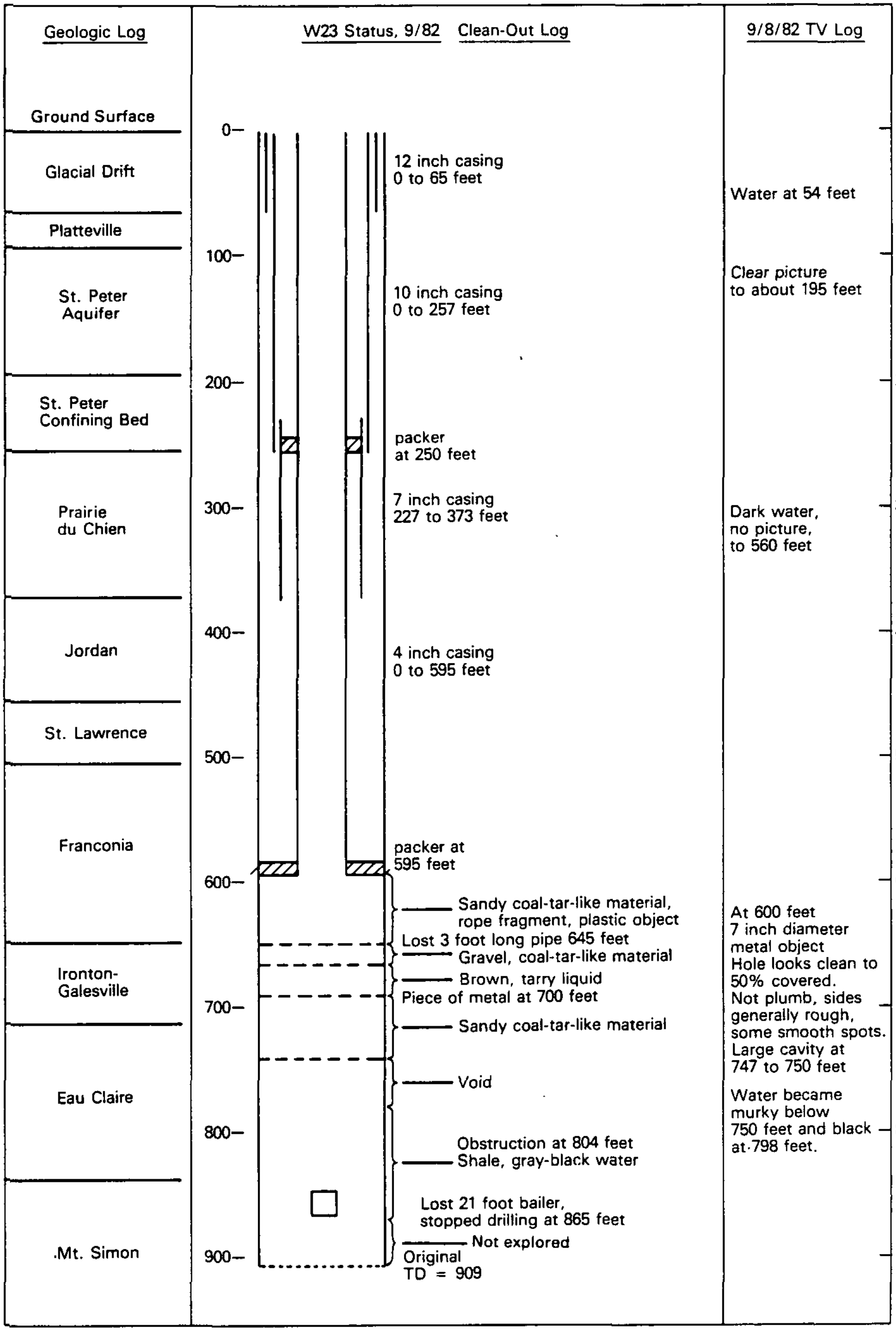

Figure D2-4 Well W23 Clean-Out Log

On July 29, 1982, the bailer being used to remove material from the well became irretrievably lodged in the hole at 866 feet (the top of the 21 foot long bailer is at 845 feet). Through August and September, attempts were made to remove the bailer. These attempts were abandoned on September 28 and bentonite pellets were used to plug the bottom of the well up to the 755 foot level. As a result of a planning meeting on October 11, more bentonite was put in W23 to the 713 foot level at the base of the Ironton-Galesville aquifer. The well bore was then cleaned by using compressed air from about 710 feet to 520 feet (Franconia, Ironton, and Galesville Formations).

This cleaning method consisted of directing compressed air (at a pressure of 200 to 300 pounds per square inch) toward the well bore using a five-foot section of 1-1/2 inch diameter pipe with four rows of 16 holes drilled along its length. The air was usually on for one-half to one hour and then shut off. In every instance, water would discharge from the well for 15 to 30 minutes after the air was shut off. The air pressure was sufficient to force water around both packers that were in the well and deliver about 280 galllons per minute. A holding pond was dug near the well so that most of the solids could settle out of the discharge, and only the clarified water was pumped to the sanitary sewer. Water samples before cleaning and after cleaning were collected and analyzed. A TV log made after cleaning showed little or no material left on the well bore.

In November 1982, bentonite was again used to plug the hole, this time to a depth of about 545 feet (middle of the Franconia Formation). At this point an attempt was made to remove the old seven-inch casing. This casing was installed when the well was built in 1918. It extended from a depth of about 230 feet to 380 feet in the well. The method for pulling the old casing involved sand-locking a four-inch pipe to the seven-inch casing and then pulling on the four-inch pipe. Compressed air was used to help loosen the casing from the well bore. On November 29, approximately 30 feet of the old seven-inch casing and almost 90 feet of ten-inch casing was removed from the well. Apparently, the seven-inch casing broke at about 260 feet where a large hole in the casing previously existed (see USGS caliper log, Figure D2-3). When this 30-foot section was raised to the surface, it caught on the section of ten-inch casing that was installed in 1933, thereby raising both casing sections to the surface.

A TV log made at this time showed an open hole from 69 to 88 feet, from 261 feet to 266 feet, and below 380 feet where the seven-inch casing ends. The last work done on W23 was the installation of pipe and packers in the well to prevent interaquifer flow (see Figure D2-5). The well was left in this condition pending additional funding for continued work on the well.

D2.3 W105 Review

W105 was located in June 1980 about 200 feet south of W23 at the plant site following about two years of research by the City of St. Louis Park, the Minnesota Department of Health, and the USGS (St. Louis Park Sun 1980). The well is plugged with dirt at the surface, and no attempts to investigate W105 in the field have been made. Therefore, the only information concerning this well is from historical records which, as was the case for W23, are sometimes conflicting and incomplete. For this reason, only a brief summary of W105 is presented based on generally agreed upon facts. Exploratory work is planned for W105 and will probably provide a much more adequate description of the well. This will enable an assessment of this well's role as a source of contamination to the bedrock aquifers.

W105 supplied at the least part of the sugar beet refinery's water supply. Therefore, it is likely that the well was drilled close to the turn of the century. When RT&CC moved onto the site, the well was cleaned out to a depth of 866 feet (Larkin 1917c). The well was probably deeper than 866 feet, but even at this depth it would have penetrated all of the bedrock aquifers under the site. The well was not used by the RT&CC plant because W23 was drilled in a more favorable location. Originally, RT&CC planned to use W105 and W23 was drilled as an alternate supply in case W105 went bad. However, because W23 was drilled next to the refinery building where the pumping equipment was housed, W105 became the alternate supply (Larkin 1917d).

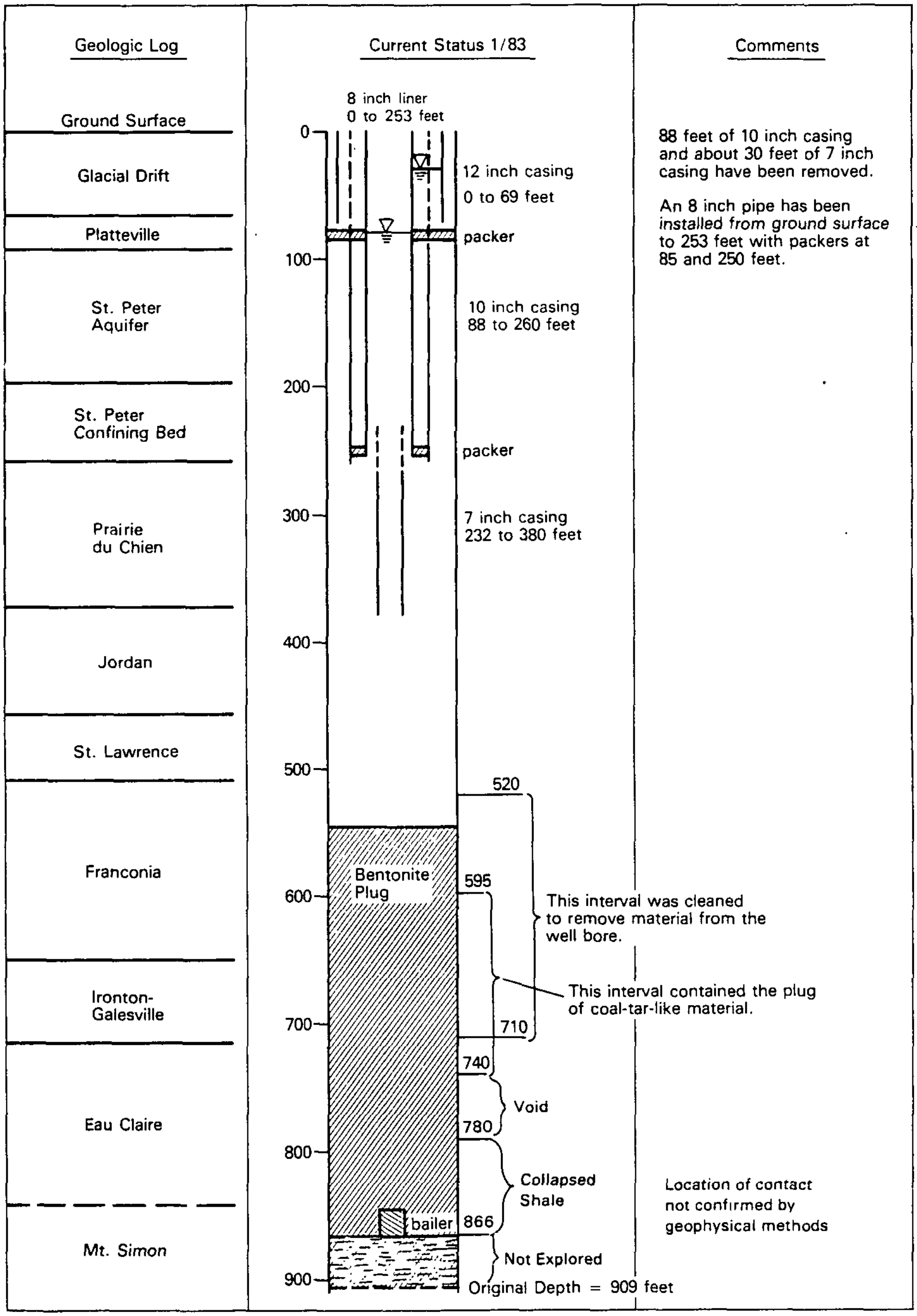

Figure D2-5 Current Sketch of Well W23

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In 1933, W105 was plugged by McCarthy Well Company at the same time that casing was being added to W23. It was McCarthy's opinion that plugging W105 and modifying W23 would improve the water quality at W112, St. Louis Park's first municipal well (McCarthy 1934). There is no further information concerning W105 in the historical record after this time.

It is likely that interaquifer flow occurred in W105 for at least a portion of its history. However, the magnitude of any impact this well has had on the water quality in any of the bedrock aquifers is unknown.

D3. REMEDIAL ACTION OPTIONS FOR W23

This section discusses the various options for remedial actions that may be taken to control W23 as a source of contamination and the associated cost of each option. In order to identify remedial action options for W23, it is first necessary to characterize the present distribution of contaminants in and around the well, and to assess the likely role of W23 as a source of contamination to each affected aquifer.

D3.1 W23 as a Source of Contamination

D3.1.1 Mt. Simon-Hinckley Aquifer

The impact that W23 has had on the Mt. Simon-Hinckley aquifer is not currently known. Water quality data from the few wells completed in this aquifer do not indicate a contamination problem (see Appendix K). However, because it is believed that the well originally penetrated 69 feet of the Mt. Simon Formation (depth of well = 909 feet, top of Mt. Simon = 840 feet), W23 may have provided a conduit to this aquifer. No evidence of any contamination was found during the recent exploratory work and the location of the top of the Mt. Simon could not be confirmed by geophysical methods. The plug of coal-tar-like material that was removed from the well in the summer of 1982 effectively isolated the Mt. Simon-Hinckley aquifer from overlying aquifers, since this plug rested on a bridge of collapsed shale in the Eau Claire Formation, about 100 feet above the Mt. Simon-Hinckley. Therefore, it is likely that W23 did not serve as a conduit for contaminant migration to this aquifer over the entire history of the well.

It is also possible that W23 provided a conduit to the Mt. Simon-Hinckley during exploration of the well this past summer. The plug of coal-tar material was removed from the well in June and July of 1982. Presumably this opened the pathway to the Mt. Simon until the well was sealed with bentonite. The time the well was open was from July 9 (when the driller broke into the void below the plug) to September 28, a total of 82 days. TV logs made during this time were obscured by black water that filled the bottom of the well. If the information on the original well log is correct, then this black water was in contact with the Mt. Simon. However, only 26 feet of the Mt. Simon was penetrated when the well was being cleaned, so it is likely that any contamination caused by the exploratory work would have little impact on the aquifer (the aquifer is about 270 feet thick).

There is not expected to be any future source of contamination in this aquifer because the bentonite plug now seals the bottom of W23.

D3.1.2 Ironton-Galesville Aquifer

The degree to which W23 has contaminated water in the Ironton-Galesville aquifer is not presently known. There are no historical water quality data for the aquifer with which to assess a contamination problem. Part of the reason these data are scarce is that very few wells use the Ironton-Galesville as a source of water. Despite the lack of measurements to identify contamination, it is probable that some contamination of the Ironton-Galesville aquifer has occurred due to W23. As shown in Figure D2-4, the plug containing coal-tar-like material was adjacent to the aquifer over its entire thickness (about 67 feet). Thus, the aquifer was directly exposed to contaminants in the plug for some unknown length of time. Direct exposure to the plug of coal-tar-like material may have created a halo of contamination around W23 in the Ironton-Galesville.

The recent exploratory work mobilized contaminants in the well and probably increased the degree of contamination in the Ironton-Galesville. After the plug of material had been removed from W23 in June and July 1982, water samples were collected over a 24-hour pumping period on October 18 and 19 from the combined Ironton-Galesville and Franconia Formations. The well bore was then cleaned using compressed air, and more samples were collected on November 4 and 5. Both sets of samples contained total PAH in the part per million range, and did not show decreased concentrations over the 24-hour pumping periods. Evidently, the pumping rate (about 20 gallons per minute) and/or the duration of the test were not sufficient to provide samples unaffected by the halo of contaminants around W23. A rough calculation of the distance of draw when pumping W23 for 24 hours at 20 gallons per minute yields only ten feet (assuming an effective porosity of 0.2 and using the method of Keely 1982). It is not known if the halo of contaminants around W23, as measured by these sampling efforts, is due to historic contact with the plug material or recent plug removal and clean-out work.

D3.1.3 Prairie du Chien-Jordan Aquifer

The degree of contamination in the Prairie du Chien-Jordan aquifer is well described by numerous water quality data, especially PAH data, collected since 1978 (see Appendices E and K). The amount of contamination contributed by W23 relative to natural percolation from surface sources and multi-aquifer flow through other wells and from sources not related to the plant is not known. However, flow into the aquifer has been measured in W23, and there is documented evidence of coal-tar-like materials in the well. Thus, it is generally assumed that W23 is a major source of contamination to the Prairie du Chien-Jordan aquifer.

While the means by which W23 contaminated the Prairie du Chien-Jordan are not completely understood, more is known about contamination from the well in this aquifer than in the other aquifers. As discussed in section D2.2, the USGS measured a flow of about 150 gallons per minute entering W23 near 215 feet through holes in the casing adjacent to the basal St. Peter Formation and exiting W23 at 264 feet through holes in the casing adjacent to the Prairie du Chien. This flow was one mechanism that transported contamination to the aquifer. Although the quality of the water in this flow was never measured, the sample taken by the USGS in 1981 and analyzed by Midwest Research Institute (MRI) may represent approximately the same water quality. However, the sample was collected almost two years after the flow had ceased. A chemical equilibrium may have been established between the ground water and coal-tar-like materials coating the well and solution channels in the Prairie du Chien. Thus, this sample is likely to contain higher PAH levels than the previously flowing water. MRI found a total of 328 micrograms per liter of PAH in that sample. At a flow rate of 150 gallons per minute this indicates that W23 was introducing about 0.6 pound per day of PAH into the Prairie du Chien-Jordan. It is not known, however, how long this interaquifer flow occurred and how conservatively the PAH concentration is overestimated.

It should not be concluded from the foregoing discussion that the formation water in the Basal St. Peter is contaminated. Instead, the most likely explanation is that contaminated water from the ground surface (during storm events) and the Drift-Platteville aquifer migrated down around the outside of the casing and entered the well at 215 feet. Note on the original well log (Figure D2-1) that a 12-inch hole was drilled from the ground surface to a depth of 258 feet. Normal well installation practices in the early part of this century did not include the placement of grout seals to prevent the flow of water in the annular space between the casing and the bore hole, and the well log does not indicate that grout was used. Therefore, it is likely that interaquifer flow from the Drift-Platteville aquifer to the Prairie du Chien-Jordan is responsible for a portion of the contamination in the Prairie du Chien-Jordan aquifer.

Other mechanisms by which W23 added to the contamination in the Prairie du Chien-Jordan are less certain. The plug of material in the well was about 140 feet below the Prairie du Chien-Jordan, and therefore the plug itself should not have been a continuing source of contamination to this aquifer. There are a variety of means by which the plug may have formed in W23, but regardless of how it got there, some fraction of the coal-tar-like material which travelled down the well remained along the well bore.

The TV log made on December 13, 1982 shows coal-tar-like material coating portions of the rock surface in both the Prairie du Chien and Jordan Formations. These deposits, at least those along the well bore, added an unknown amount of contaminants to the aquifer, and probably created a zone of contaminated water and aquifer materials around the well. It is not clear at present if the material in the solution channels of the Prairie du Chien was deposited there by contaminated water or if that material was carried there by interaquifer flow as a separate phase. Similarly, it is not known if the deposits in the solution channels act as continuing sources of contamination, or if the material is irreversibly adsorbed.

A third mechanism for contamination to this aquifer occurred during the recent work done on the well. From October 21 to October 29, 1982 the well bore from 710 to 520 feet was cleaned using compressed air. The process involved using air under a pressure of 200 to 300 pounds per square inch to clean five-foot increments of the well bore, taking about forty-eight cycles to clean the 190 foot interval in the well. As originally set up, all the discharge from this operation was to be directed to the surface through a four-inch pipe and then to the sewer. In practice, the air pressure was too great for the packers in the well and forced a portion of the discharge to the surface in the annular space between the four-inch pipe and ten-inch casing. After a minute or two into the cleaning cycle, the discharge from the ten-inch casing would stop, and all the remaining flow came from the four-inch pipe. The water level in the ten-inch casing receded to about 40 feet after flow had stopped. Thus, the volume of water that receded back into W23 was the volume of the top 40 feet in the annular space between the four-inch and ten-inch pipes, or about 140 gallons. The water that receded back into the well must have entered one or more of the aquifers penetrated by the well (including all but the Mt. Simon-Hinckley aquifers) but it is not known exactly where this water went. The concern is that the water was highly contaminated, based on visual inspection*, and that this water was released to at least one of the aquifers penetrated by the well, thus adding to contamination in that aquifer.

*Samples of this water were collected in an Imhoff flask at various times during the cleaning operation. An estimated 10 to 20 parts per million of separate phase coal-tar-like materials were in these samples. The total amount of contaminants removed during the forty-eight cycles of air cleaning are estimated to be one pound.

Future contamination of the Prairie du Chien-Jordan aquifer due to W23 could occur by means of 1) coal-tar-like material coating the well bore and solution channels of the Prairie du Chien, and 2) a halo of contaminated ground water and aquifer materials in the Jordan Sandstone. It is not known how the magnitude of these future sources of contamination compare to historic sources or to sources other than W23.

D3.1.4 St. Peter Aquifer

W23 may have contributed some unknown degree of contamination to the St. Peter aquifer as a result of leakage of Drift-Platteville water around the ungrouted ten-inch casing. There have been no measurements made in W23 or in any other well to explore the possibility. As discussed in the previous section, the recent air cleaning in the well may have caused some additional contamination in the St. Peter.

A possibility exists for continuing contamination from W23 if a halo of contaminated ground water and/or aquifer materials exists around the well, or if there are coal-tar-like materials deposited on the outside of the ten-inch casing. The magnitude of any of the past, present, or future sources is unknown.

D3.1.5 Drift-Platteville Aquifer

The relative impact that W23 has had on the water quality in the Drift-Platteville aquifer system is likely to be small compared to contamination received by this aquifer from a variety of sources directly from the plant (see Appendices A and B). W23 was located just outside the southeast corner of the refinery building in an area that might have been subject to relatively high contaminant loadings on the surface. In any case, W23 is not an important future source of contamination to the Drift-Platteville aquifer.

On the other hand, there is reason to believe that the Drift-Platteville aquifer was at least in part responsible for contaminants entering W23. Before 1933, W23 was not cased adjacent to the Platteville Formation (65 to 80 feet), so it is likely that there was interaquifer flow from the Platteville via W23 to deeper bedrock aquifers. The degree to which the upper aquifer was contaminated prior to 1933 is not known; however, it is conceivable that this mechanism was adding to contamination in and around the well. When casing was added to the well in 1933, leakage in the annular space between the casing and the well bore probably continued to contaminate deeper aquifers. Note that the holes in the ten-inch casing around 215 feet would have allowed this leakage to re-enter, and thus contaminate, lower portions of the well.

D3.2 W23 Source Control Alternatives

Before discussing future source controls for W23 it is appropriate to first review the previous remedial work done in the well, and to then recognize limitations that must be imposed on future controls due to ongoing exploratory work that must preceed remedial work. Major remedial actions done to date in W23 include: removal of the plug of coal-tar-like material, removal of coal-tar-like material coating portions of the well bore from approximately 520 to 710 feet below ground surface, and the installation of a bentonite plug from the bottom at 830 feet to about 545 feet. These remedial actions serve to reduce the amount of materials acting as contamination sources to the Ironton-Galesville and Mt. Simon-Hinckley aquifers, and eliminate the possiblity of interaquifer flow to these aquifers.

The actual effectiveness of the previous remedial work is difficult to assess. One difficulty in this regard is that water quality data were not available, or were not collected before work started in the well. Therefore, water quality data cannot be used to quantitatively assess the effectiveness of remedial work. TV logs made before and after the remedial work provide visual, qualitative assessments of the work, however, it is doubtful that this technique is adequate given the importance attached to even low concentrations (part per trillion) of contaminants. The single exception to this lack of data is the sample collected in 1981 by the USGS and analyzed by Midwest Research Institute. The sample probably represented water in the Prairie du Chien-Jordan aquifer, and MRI found 328 micrograms per liter of total PAH in the sample. The 1982 work on W23 did not include taking samples from the Prairie du Chien-Jordan aquifer.

Another difficulty encountered in trying to assess the effectiveness of remedial work done on W23 involves uncertainties regarding the transport phenomena associated with coal-tar-like materials in the well. For example, it seems beneficial to have removed the plug of coal-tar-like material from the well, because the material removed is no longer a threat to contaminate any ground water. However, there is no way to quantitatively determine if all of the plug was removed. If the surface area of the coal-tar-like material is the major factor controlling the rate of contaminant release, then it is not clear what benefit is derived from removing less than 100% of the mass of material. Similarly, the rate at which contaminants from materials coating various casing segments and/or rock surfaces in the open well bore migrate into the ground water may not be significant compared to the effects of trying to remove this material, owing to the probable long length of time the coatings have been there.

Another problem that inhibits the evaluation of remedial work done on W23 involves two complications brought about by the use of compressed air to clean portions of the well bore. As discussed in section D3.1.3, about 140 gallons of contaminated water receded back into W23 with each air cleaning cycle, and it is not known what effects this water has had on any of the aquifers. In addition, the compressed air may have forced a portion of the contaminants out into the formation (mostly into the Ironton-Galesville, but the Jordan could be affected too) instead of to the surface. Therefore, it may not be possible to accurately assess the effects of any of the previous remedial work in W23. Future assessments using water quality data as a basis must take into account these uncertainties.

Plans for remedial work in W23 should recognize that this well probably cannot be used to reliably control sources in all affected aquifers because 1) the Mt. Simon-Hinckley is not readily accessible in W23, and 2) the ten-inch casing can only accommodate a limited amount of down hole equipment (such as pumps and pipe).

Based on the previous discussions, W23 has been, and will continue to be, a source of contamination to each of the bedrock aquifers it penetrates. However, contamination of the Mt. Simon-Hinckley appears to have only occurred during the work in the summer of 1982. This contamination was very slight and any further addition of contaminants is prevented by the bentonite plug. Because the magnitude of contamination from W23 as compared to other sources is not known for each of the aquifers, the following sections offer alternatives to control W23 as a source in each aquifer. Actual recommendations for the use of W23 to control contamination in any of the aquifers must be based on the overall strategy used to meet the objectives of this study. For example, while alternatives are presented for controlling W23 as a source in the Ironton-Galesville aquifer, it is not clear that any controls should be implemented here because the Ironton-Galesville is generally not used as a ground-water resource in the Twin Cities area.

D3.2.1 Mt. Simon-Hinckley Aquifer

Alternatives for controlling W23 as a source in the Mt. Simon-Hinckley, and all the other aquifers, differ with respect to how reliably they control the source. The reliability of control alternatives is also affected by the unknown character of the source. In the Mt. Simon-Hinckley, the very existence of a source is questionable. Thus, the source control now in place, the bentonite seal in the bottom of W23, may well be effective.

A reliable source control for the Mt. Simon-Hinckley would be a pumping well close to W23. This would serve to control the spread of contaminants, as well as to remove contaminants from the aquifer. Well W23 cannot be used for this purpose because of the bailer lodged in the well, thus the alternatives considered for this purpose include drilling a new well close to W23 or using W105. A new well drilled close to W23 is the superior choice, because W105 is too far from W23 to be effective. A new well that is drilled close to W23 would need to be pumped at about 100 gallons per minute. Disposing of contaminated water is the most important factor controlling the cost of the W23 pumping alternatives (See Appendix F). Using the disposal scheme of discharging to the sanitary sewer as outlined in Appendix F, the present value cost of disposing 100 gallons per minute for 100 years is about $750,000.

A sampling program with subsequent water quality analyses would be part of any pumping alternative for W23. The rationale for this program is that, after a period of years, it is possible that the PAH concentrations would decrease to a point where no further pumping would be required. Pumping at 100 gallons per minute for 50 years instead of 100 years would reduce the present value cost of disposing the discharge from $750,000 to $670,000. If pumping would stop after only 20 years, the present value costs of disposing the discharge would be $460,000.

For either pumping alternative, the water disposal costs (for 100 years at 100 gallons per minute) along with drilling, equipment, sampling and analytical costs would result in a total cost of about one million dollars. The alternative of sealing W23 to prevent further interaquifer flow with the Mt. Simon-Hinckley is already in place and has no further costs associated with it.

D3.2.2 Ironton-Galesville Aquifer

The alternatives for source control in W23 for this aquifer closely parallel the alternatives considered for the Mt. Simon-Hinckley aquifer, with the difference that W23 itself could be redrilled and used as a pump-out well. If W23 was used for this purpose, it is doubtful whether or not source controls could be implemented in W23 for other aquifers. Therefore, this alternative must be evaluated in light of meeting the overall objectives of this study. Although previous water quality analyses have shown that the Ironton-Galesville aquifer is contaminated in the vicinity of W23, it is not known for how long or at what rate the aquifer must to be pumped in order to contain or remove contaminants. For this reason, the disposal costs are again the major factor which determine the total cost of pumping alternatives. As with the Mt. Simon-Hinckley, one million dollars is the total estimated cost for any of these alternatives. On the other hand, a bentonite seal preventing interaquifer flow is already in place and the well bore has already been cleaned. These actions have no additional costs in the reasonably forseeable future.

If a sampling and analysis program could shorten the pumping duration, then the present value cost of the pumping alternatives would decrease as described in section D3.2.1.

D3.2.3 Prairie du Chien-Jordan Aquifer

Source control alternatives in W23 for this aquifer are considered to have priority over those for other aquifers, because of the importance of the Prairie du Chien-Jordan as a drinking water supply resource. However, due to uncertainties regarding the past, present and future role of W23 as a source of contamination to the aquifer, the reliability, effectiveness, and therefore the appropriateness of source control alternatives are likewise uncertain. Thus, the emphasis must be placed on working to integrate W23 source controls with other control measures to satisfy the objectives of this study.

Source control alternatives for the Prairie du Chien are categorically the same as for the two deeper aquifers previously discussed, namely, grouting or pumping. There is more flexibility for alternatives in this aquifer because W23 has not been sealed in this aquifer and further exploratory work will likely take place before final controls are used at the well.

Grouting Alternatives

Sealing W23 with grout (bentonite, cement, etc.) is an attractive source control alternative because there are no water disposal costs associated with it. Grouting would effectively control the interaquifer flow of water into the Prairie du Chien-Jordan, which appears to have been the major historical contamination source. A decision to grout the well must acknowledge the fact that contaminated ground water and aquifer materials left in place in and around the well will continue to act as a source in the future. Thus, such a decision would be based on the conclusion that contamination from these sources inside and immediately outside of W23 are negligible compared to 1) past sources, particularly interaquifer flow, 2) other sources, and 3) the effects of eliminating these sources on potable water supply for St. Louis Park and surrounding communities and on the use of ground-water resources in the area.

Grouting the well in its current state would cost about $10,000 to $20,000 based on $1,000 per day for the driller and all necessary equipment, materials and labor. It is not known how much difficulty the Prairie du Chien solution channels will pose to installing a grout plug, but the driller should be able to complete the job in 10 to 20 days given that the previous bentonite seals were installed rather quickly.

A variation of this grouting alternative would be to first remove the seven-inch casing in the well and view the entire interval of Prairie du Chien with a TV camera to assess the extent of contamination. This step is intended to increase the reliability of grouting W23. Pulling the casing would cost about $5,000 to $10,000. Grouting without the casing would be more difficult due to the presence of more solution channels. The total cost for this alternative is estimated to range from $15,000 to $30,000.

Another grouting option would be to clean the well bore in the interval adjacent to the Jordan sandstone. Then the only continuing source of contamination would be the halo of contaminants around the well and any material in solution channels that did not get sealed by grout. The effectiveness of the well bore cleaning operation is unknown, and it may be that more contaminants are mobilized in the short term via this process rather than removed. The cost of well bore cleaning is estimated to be about $5,000, increasing the total cost of this option to between $20,000 and $35,000.

Pumping Alternatives

Pumping W23 with a packer installed to isolate the Prairie du Chien-Jordan aquifer is an alternative for controlling the migration of contaminants from within W23 and of contaminated ground water up to hundreds of feet away from W23. As with the grouting alternatives, various combinations of pumping with well cleaning, casing removal, and even limited grouting are possible.

The first pumping alternative considered is pumping W23 at 400 gallons per minute to control a large volume of the Prairie du Chien-Jordan aquifer. This alternative offers the maximum reliability for W23 source controls. First, the seven-inch casing would have to be removed so that it would not interfere with the flow of water to the well. At that point a TV log could be used to visually inspect the extent of contamination in the Prarie du Chien and any other exploratory work could be performed. The major cost associated with these activities is disposal of the well discharge. At 400 gallons per minute, it would be more cost-effective to treat the discharge and send it to Minnehaha Creek than to send it untreated to the sanitary sewer (see Appendix F). A pumping rate of 400 gallons per minute would capture water within at least 1000 feet in all directions from W23 (based on ground water model results). However, the discharge disposal cost alone would be around 1.6 million dollars (see Figure F5-1 in Appendix F). The cost of removing the casing, TV logging, and other down-hole work that may be done (i.e., cleaning the well bore, grouting to the bottom of the Jordan, installing a pump and packers, etc.) is estimated to be about $50,000.

Another alternative is to pump W23 at a much lower rate in order to control contaminants within only about 50 feet of the well. It is likely that a halo of contaminated ground water and aquifer materials exists around W23, but the extent of this halo is not known. In order to control this source, the area to be controlled must first be determined, then the appropriate pumping rate must be calculated. Both of these items could probably be addressed in further exploratory work at the well in the form of a pumping test with time series water quality sampling. It is estimated that a pumping rate of 50 to 100 gallons per minute will be appropriate. The total costs for this range of pumping is about one-half to one million dollars. Again, the discharge disposal costs are the major factors which determine this total.

In summary, source control alternatives for W23 in the Prairie du Chien-Jordan aquifer include grouting and pumping options, of which the pumping options exceed the cost of grouting by more than factor of ten. Grouting alternatives cost on the order of $10,000 to $35,000, depending on whether the seven-inch casing is pulled and the well bore is air cleaned. There are no subsequent operation and maintenance costs associated with grouting alternatives. Pumping alternatives have a present value cost of from 0.5 to 1.7 million dollars, depending on the flow rate, and must be operated for years. The choice between the two options is dependant on how effective the source controls need to be, as determined by the possible effects of a continuing source of Prairie du Chien-Jordan contamination at W23 on the effectiveness and cost of drinking water treatment or alternate drinking water supply alternatives.

D3.2.4 St. Peter Aquifer

Contamination in the St. Peter aquifer due to W23 has not been documented, although leakage around the casing may have contributed contamination over the history of the well. A packer now prevents leakage from the Platteville. The possibility of continued contamination exists if material has been deposited in a halo around W23 or on the outside at the ten-inch casing. If this source warrants control, the following options are available:

> Option 1: Drill a new well near W23 to determine the degree of contamination in the St. Peter. If no contamination is found, then the total cost is estimated at $25,000. If contamination is found, the new well could be pumped to control this source. Although the pumping rate could be kept low (on the order of 20 to 50 gallons per minute) the disposal cost would bring the total cost of this option to

about $200,000 (present value for 100 years - see Appendix F). Note that W23 could also be used for this pumping option, however, the modifications to W23 coupled with disposal costs would result in approximately the same total cost. Depending on the controls used elsewhere in W23, it is likely that the new well option would be more satisfactory.

Option 2: Remove the ten-inch casing to eliminate the possibility of a continuing source of contamination from material on the outside of the casing. It is questionable whether the drillers could do this and maintain an open hole due to the loose structure of the St. Peter sandstone. This activity could interfere with other remedial actions taken at the well, and would not address the question of the extent of contamination in the aquifer. A seal would have to be made between the Platteville and the St. Peter to prevent interaquifer flow. The total cost of this option is estimated at $10,000 to $20,000.

Option 3: Grout this section of the well (95 feet to 260 feet) to prevent interaquifer flow. Grouting would not address contamination outside the ten-inch casing. The total cost of this option is estimated to be $5,000 to $10,000.

D3.2.5 Drift-Platteville Aquifer

As previously discussed, W23 is not believed to contribute significant contamination to the Drift-Platteville aquifer, although the reverse may be true. If, however, there was a need to control W23 as a source in this aquifer as part of the overall remedial actions, then the twelve-inch casing could be removed from the well. The casing is likely to be partially coated with coal-tar-like material, therefore its removal would also remove some contaminants from contact with the aquifer. This work would probably conflict with other remedial work in the well. Any other source controls for the Drift-Platteville aquifer would not be focused on W23.

D4. OFF-SITE MULTI-AQUIFER WELLS

D4.1 Well Inventory

The primary data source for ERT's assessment of off-site multi-aquifer wells has been a USGS inventory of multi-aquifer wells completed by Hult and Schoenberg (1981) for the two-square-mile area surrounding the site. This inventory was compiled from files of the USGS, the Minnesota Geological Survey, the St. Louis Park Department of Public Works, and previous reports. These data were augmented by discussions with area residents, employees of local businesses, and drillers (Hult and Schoenberg 1981).

The USGS well inventory is presented in tabular form, indicating for each well: a designated well number, latitude and longitude, driller, date drilled, driller's log, land surface altitude, reported depth of well, casing schedule, aquifers open to well bore, water level, date measured, and field measurement status (whether located, used, sealed, logged, destroyed, etc.).

Other ERT data sources include Minnesota Department of Health (MDH) and USGS memos, geophysical and television logs and interpretations, and reports by Sunde (1974), Barr (1977), and Hickok (1981).

Hickok has recently finished a well survey in a four-square-mile area around site (Figure D4-1). Hickok's report has not yet been made available to ERT but we understand that Hickok has identified approximately 1,000 wells in the immediate site area (Hansel 1983). The vast majority of these wells are reported to be shallow (i.e. penetrating no deeper than the Drift-Platteville), with occasional deeper wells (i.e., penetrating the St. Peter). Some wells, usually associated with indistrial users, penetrate the Prairie du Chien or deeper. ERT also understands that the USGS has been in the process of updating their 1981 well inventory. However, until ERT can review these documents no conclusions can be drawn from them.

The data base used by ERT for our study and discussion of multi-aquifer wells is limited to the U.S. Geological Survey inventory (Hult and Schoenberg 1981) a U.S. Geological Survey letter from

NON-RESPONSIVE

[redacted]

M. Hult (1979) to the MDH, and previous reports (Sunde 1974; Barr 1976 and 1977; Hickok 1981). ERT did not conduct door-to-door surveys or file searches to augment existing data sources, nor did ERT have access to the on-going Hickok well inventory or the up-dated USGS inventory.

Listed in Table D4-1 are multi-aquifer wells identified by the U.S. Geological Survey and Minnesota Pollution Control Agency. Table D4-2 presents other wells listed in the USGS well inventory which possibly penetrate more than one aquifer.

D4.2 Multi-aquifer Wells and Contaminant Movement

D4.2.1 Multi-aquifer Well Hydraulics

Multi-aquifer wells provide a means for ground water to circumvent natural patterns of flow, resulting in hydraulic connection between aquifers which are separated by effective confining strata. Interaquifer flow through multi-aquifer wells is largely controlled by differences in potentiometric surface between aquifers. Within the study area, the potentiometric surface is higher in the uppermost confined aquifer, decreasing progressively in the underlying aquifers; thus, interaquifer flow is in a downward direction.

Multi-aquifer wells provide several mechanisms for interaquifer flow. During open-hole construction, water may flow freely down the hole from top to bottom, intersecting various aquifers. In wells that are cased after construction, this phenomenon may continue over the life of the well, unless the aquifers are isolated by grouting. Even in situations where casing is installed over the entire depth of the well, leaks in the casing may permit some interaquifer flow. This situation may become more pronounced over time due to corrosion of the casing. Additional interaquifer flow may occur within the annular space between the casing and the well bore.

The volume of interaquifer flow through multi-aquifer wells may be substantial; interaquifer flow from the Drift-Platteville and St. Peter aquifers into underlying aquifers via multi-aquifer wells has been estimated to exceed the amount of water withdrawn from these

TABLE D4-1

MULTI-AQUIFER WELLS IDENTIFIED BY USGS, MDH AND MPCA

Well Number	Well Name	Aquifers Open to Well[a]: Drift-Platteville	St. Peter	Prairie du Chien-Jordan	Ironton-Galesville	Mt. Simon-Hinckley	Status[b]	Source[c]	Remarks
Deep Wells									
W29	Flame Industries	X?	X?	X?			P	H,M,U	1976 in minor use
W32	Texatonka Shopping Ctr.			X			L	H,U	
W34	Crib Diaper Service			S			SM,G	H,U	
W35	Burdick Grain Co.	X?	X?	X			MW;P	M,H,U,MPCA	
W38	Milwaukee RR Well	S	S	S	X	S	MW	H,U,UE	
W40	Minnesota Rubber			X			P	H,M,U	1976 in active use
W45	S&K Products, Inc.		X	X			P	H,M,U	
W46	S&K Products, Inc.		X	X			P	H,M,U	1976 in minor use
W47	Belco; Burdick Grain	S		S			S;G	U,H	
W49	Strom Block Co.		X	X			-	H,M,U	
W50	Prestolite		S	S	S		SM;G	U	
W62	McGourtney Plastics		X	X			P	H,M,U	
W66	Black Top Service	X?	X?	X?			RF	H,U	deep well
W69	Hedberg-Friedheim	S	S	S			SM;G	H,U	Wolfe Lake
W70	Park Theatre			X			P	H,U	
W74	Landers Gravel	X?	X?	X			RS	S,U	
W105	Minn. Sugar Beet	X	X	X		X?	-	H,U	under study by Hickok, 1983
W107	Interior Elev.	X?	X	X	X		-	H,U	
W112	Old SLP #1			X			MW;G	H,U	Old St. Louis Park Well SLP #1
W114	Hedberg-Friedheim	S	S				SM	U	

TABLE D4-1 (Continued)

Well Number	Well Name	Aquifers Open to Well[a] Drift-Platteville	St. Peter	Prairie du Chien-Jordan	Ironton-Galesville	Mt. Simon-Hinckley	Status[b]	Source[c]	Remarks
Shallow Wells									
W27	Terry Excavating	X	S				MW;G	H,U	11 Oct 79 St. Peter sealed
W30	3636 Quebec	S	S				SM	H,U	
W33	Strand Mfg.	S	X				P; MW; G	H,U	11 Oct 79 Platteville sealed
W37	Dayton Rogers #2	X	X				MW;G	H,U	
W41	Hartman #1	X	X				O	M	
W44	Kings Inn	X	X				P	H,U	
W52	Merit Gauge	S	S				SM; G	H,U	
W60	3645 Rhode Island	S	S				SM	H,U	
W61	W.V. Terry						-	H,U	No Data
W65	Ace Mfg.	X	X				L	H,U	
W67	Black Top Service	X?	X				-	H,U	Shallow Well
W75	Park Pet Hospital	X	X				P	H,U	
W76	Professional Instru.	X?	X				P	H,U	
W106	Hedberg-Friedheim	X?	X				-	H,U	
W113	SLP #3	X	X				P	H,U	St. Louis Park Well SLP #3

(a)Aquifer Legend
X, aquifer presently open to well; X?, aquifer probably open to well; S, aquifer no longer open to well.
(b)Status Legend
D, destroyed; O, obstructed; L, located; P, located with pump; S, sealed; RS, reportedly sealed; SM, sealed by MDH; MW, reconstructed as monitoring well; RF, reportedly filled; G, geophysically logged.
(c)Source Legend
H, Hult (1979); U, Hult and Schoenberg (1981); M, Minnesota Department of Health (1979-1981); UE, USGS-ERT meeting of April 1980; MPCA, Minnesota Pollution Control Agency (1982).

TABLE D4-2

OTHER POSSIBLE MULTI-AQUIFER WELLS

Well Number	Well Name	Aquifers Open to Well: Drift-Platteville	St. Peter	Prairie du Chien-Jordan	Ironton-Galesville	Mt. Simon-Hinckley	Status[a]	Source[b]	Remarks
W39	3612 Alabama		X				D	U	
W48	Methodist Hospital		X	X			P; G	U	also screened in St. Lawrence Form. 285' deep
W63	National Foods			X			P	U	
W72	Harder Res.		X				-	U	
W73	Jasperson Dairy	X	X				-	U	
W80	Red Owl			X			-	U	
W82	Weldwood Nursing	X?	X?	X?			-	U	
W86	Prudential #1			X			-	U	also screened in St. Lawrence Form.
W104	Rice Sand & Gravel			X?			-	U	
W109	Max Renner's Shop	X?	X				-	U	
W111	6030 Oxford		X				L; G	U	
W118	Golf Course			X			RS	U	also screened in St. Lawrence Form.
W119	Golf Course			X			-	U	also screened in St. Lawrence Form.

(a) Status Legend
D, destroyed; O, obstructed; L, located; P, located with pump; S, sealed; RS, reportedly sealed; SM, sealed by MDH; MW, reconstructed as monitoring well; TS, temporarily sealed; RF, reportedly filled; G, geophysically logged.

(b) Source Legend
H, Hult (1979); U, Hult and Schoenberg (1981); M, Minnesota Department of Health (1979-1981); UE, USGS-ERT meeting of April 1980.

two aquifers in the vicinity of the site (Hult and Schoenberg 1981). Interaquifer flows of this magnitude have a noticeable effect on the local potentiometric surface, creating a cone of depression in the overlying aquifer and a corresponding cone of impression in the underlying aquifer.

The patterns of interaquifer flow through multi-aquifer wells are important, since contaminated ground water may follow these pathways. Factors influencing the rate and direction of interaquifer flow include: well design and construction, present well condition, formation permeability (including local effects of plugging or encrustation), differences in water level between aquifers, and effects of pumping nearby wells. Certain conditions, such as encased wells, severely deteriorated casing or high formation permeability (e.g. presence of solution cavities) in the portion of the aquifer open to interaquifer flow, favor relatively rapid movement of contaminated ground water in multi-aquifer wells.

Interaquifer flow in a given well is typically difficult to assess since the condition of the well is unknown. The condition of the well affects the ability of the well to convey flow and may also create more connections from the aquifers to a cased well. Deteriorated casing and caving of the well bore are common in older wells. Flow from the aquifer may be affected by localized clogging of the aquifer pores due to sediment accumulation or chemical encrustation. These many potential factors complicate interaquifer flow hydraulics and make difficult prediction of flow.

D4.2.2 Multi-aquifer Wells in St. Louis Park

Multi-aquifer wells are believed to be significant contributors to contamination problems in St. Louis Park (Hult 1979 and Hult and Schoenberg 1981). The importance of multi-aquifer wells relative to on-site wells and other sources is not known; however, planned, current and previous mitigative actions by Minnesota state agencies include multi-aquifer well abandonment. The following is a discussion of likely multi-aquifer well effects in each aquifer.

Mt. Simon-Hinckley

Due to its great depth, few wells penetrate the Mt. Simon-Hinckley. Three multi-aquifer wells to the Mt. Simon-Hinckley have been identified within the study area: W23 and W105 on the plant site and W38, the abandoned Milwaukee Railroad well. Wells W23 and W105 are discussed in sections D2 and D3 of this appendix. Well W38 has been reconstructed as a monitoring well and no longer acts as a multi-aquifer flow path. Because of its great depth, additional multi-aquifer wells to the Mt. Simon-Hinckley are unlikely. Additional deeps wells are probably restricted to commercial and industrial areas since residential wells would not be drilled to the depth of the Mt. Simon-Hinckley. Further remedial actions for Mt. Simon-Hinckley multi-aquifer wells other than W23 and W105 are thus likely to be very limited in scope and relatively inexpensive. Depth determinations for industrial and commercial wells found in the recent Hickok survey is a sufficient action in this respect, with possible well abandonment of any additional Mt. Simon-Hinckley wells discovered. In addition, the area of the survey should be expanded to search for Mt. Simon-Hinckley wells in areas of present or likely future contamination in the Prairie du Chien-Jordan.

Prairie du Chien-Jordan

Multi-aquifer wells to the Prairie du Chien-Jordan occur more frequently than do those to the Mt. Simon-Hinckley. Travel of contaminant from the St. Peter aquifer to the Prairie du Chien-Jordan is of unknown significance. Wells which monitor only the St. Peter, and do not also connect to the Drift-Platteville, are sparse. Thus, there are limited data to define possible contamination in the St. Peter. Better definition of possible contamination in the St. Peter is a necessary step to determine the role of St. Peter to Prairie du Chien-Jordan interaquifer flow as a pathway for contamination.

Multi-aquifer wells may also convey flow from the Drift-Platteville to the Prairie du Chien-Jordan. This is of greatest concern in those areas where the upper aquifer is know to be contaminated, as mapped in Figure D4-1. Known wells in this area which connect to the Prairie du Chien-Jordan aquifer are W23, W50 and W105. Of these, W23 is temporarily sealed pending further exploratory work and W50 has been sealed. Given the intensity of past investigation of this area, it is unlikely there are many additional wells which have not yet been discovered.

St. Peter

A great many wells in the St. Louis Park area are open to both the Platteville and St. Peter. Location of all potential multi-aquifer wells in this category is very difficult due to the large number of wells involved. This is indicated by the report that Hickok's recent multi-aquifer well survey identified about 1,000 wells in a four-square-mile area bounded by West 28th Street to the north, Virginia Avenue to the west, France Avenue to the east, and Excelsior Boulevard and West 40th Street to the south. About half of these wells had not been identified before. Most of these 1,000 wells are shallow wells, although a significant number are reported to extend as far as the St. Peter (Hansel 1983). Tables D4-1 and D4-2 indicate the 25 wells connecting the Drift-Platteville and St. Peter aquifers known to this study.

The USGS (Hult and Schoenberg 1981) has indicated that shallow multi-aquifer wells do not necessarily show flow from the Platteville to the St. Peter. They postulate that chemical encrustation in the Platteville prevents flow. In any event, the possibility that such wells do not allow interaquifer flow and the large number of such wells imply that further investigation would be prudent before embarking on an expensive program to seal all such wells. Selective reconstruction of a number of wells as St. Peter monitoring wells is a cost-effective means to determine the extent of contamination in this aquifer without the expense of drilling new monitoring wells.

The impetus to determine ground-water quality in the St. Peter is the indication from the limited available data that the aquifer may be relatively uncontaminated. One such indication came with the abandonment of well W33 (Strand Manufacturing). Prior to reconstruction, the well connected the Platteville and St. Peter. Based on phenolic concentrations, the well was clearly contaminated. After reconstruction as a St. Peter monitoring well, low PAH levels were measured (the concentration of total PAH based on thirteen compounds was 11 nanograms per liter) implying that the previously measured phenolic contamination was due to Platteville water and not St. Peter water. Another indication is the failure by Hult and Schoenberg (1981) to detect multi-aquifer flow from the Drift-Platteville to the St. Peter in uncased wells which they logged geophysically and with television. Thus, although there is reason to suspect that contamination of the St. Peter may be minor, additional information is required to confirm this.

D5. MULTI-AQUIFER WELL REMEDIAL ACTIONS

D5.1 Monitoring Option

The basis for the monitoring approach is that remedial action may not be necessary for some or all multi-aquifer wells. With no action other than monitoring, multi-aquifer wells may continue to allow interaquifer flow of potentially contaminated ground water from shallower aquifers (Drift-Platteville, St. Peter) to deeper aquifers (St. Peter, Prairie du Chien-Jordan, Mt. Simon-Hinckley). No action will be taken to locate or investigate previously unidentified multi-aquifer wells.

The implications of adopting a monitoring approach depend largely on the rates of ground-water flow in the affected shallower aquifers and the ability of contaminants to reach the deeper aquifers by means of interaquifer flow through multi-aquifer wells. Once released through the well bore to lower aquifers, contamination is governed primarily by the rate and direction of ground-water flow in the lower aquifers.

There are no costs associated with the monitoring option, other than continued monitoring cost, which are discussed in Appendix J.

D5.2 Permanent Abandonment/Reconstruction/Replacement Option

Previous studies have identified many multi-aquifer wells in the study area. The recently completed Hickok inventory has apparently identified more such wells (Hansel 1983). Several multi-aquifer wells previously identified as allowing interaquifer flow have been permanently abandoned by the Minnesota Department of Health (Hult and Schoenburg 1981). The purpose of this section is to outline procedures for permanent well reconstruction, abandonment and replacement if necessary.

Within areas where there is known contamination of surficial aquifers, it serves little purpose to log multi-aquifer wells prior to permanent abandonment, since, under this option, all multi-aquifer wells in areas of known contamination are to be permanently sealed.

Flow meter logs cannot prove conclusively that low velocity interaquifer flow does not exist within the well bore. In addition, a flow meter cannot measure interaquifer flow which occurs in the annular space between the casing and the well bore.

As part of this option, multi-aquifer wells can be abandoned according to the following general guidelines. After any obstructions in the well are removed, the well can be sealed with cement grout pumped through a tremie pipe (that is, pumped from the bottom of the hole). The grout will be pumped until it is observed flowing out of the casing at the ground surface. If solution cavities or extensive fracture systems exist, casing may be required to seal them. Finally, soil will be excavated around the well head at the surface to allow a thick cement seal to be installed around the casing.

Any multi-aquifer wells that remain in good condition can be considered for reconstruction. Multi-aquifer wells can be reconstructed into single aquifer wells by sealing off portions of the well bore in all but one aquifer. In cases where the deeper aquifers are to be sealed, cement grout can be pumped into the well through tremie pipe until the grout level reaches the shallowest aquifer. In situtations where the shallower aquifers are to be sealed, new casing can be installed and grouted in place. Once the grout is set, the well can be redrilled below the new casing. Combinations of techniques can be applied to special situations (i.e. middle aquifer to be screened or left open, shallow and deep to be sealed). Reconstructed wells can be used for monitoring purposes, or water supply.

Reconstructed wells often yield less water than the original well. Several factors account for the reduced production. Following reconstruction, only one aquifer remains open to the well bore. This can result in a lower yield. In addition, reconstruction could require smaller casing to be installed. Thus, reconstructed wells may experience reduced yield due to the reduction of pump size required to fit the new casing. Actual reductions would be based on aquifer characteristics and pumping rates, and would probably only affect large volume commercial and industrial users.

Wells which are currently inactive and require abandonment will not be replaced. Active wells requiring abandonment will be replaced on the basis of their original capacity. Large capacity (high yield) wells for commercial or industrial use will be replaced with new single aquifer wells drilled to the shallowest aquifer with acceptable quality for the type of use (i.e., process or potable). Abandoned small capacity (low yield) wells for domestic, or light commercial or industrial use will be replaced with municipal water supply. Operational but seldom-used wells will not be replaced. In those situations where no other water supply is presently online, municipal water service will be tied in.

Costs

Table D5-1 presents cost information for multi-aquifer well abandonment, reconstruction, and replacement. These costs are conservative in that they assume that a drill rig will be required to clean out the well and install the grout seal. In cases where the well to be sealed is in good condition, grout can be installed by a two-person crew at substantially less than the drill rig cost of $100.00 per hour for rig and crew. However, in most cases, some clean-out work will require drill rig assistance.

Grout costs reflect the large volumes of grout which may be required to fill small cavities in the St. Peter Formation and small solution channels in the Prairie du Chien Group. Large voids would necessitate grouting around casing installed to the appropriate depth.

In cases where the multi-aquifer well to be sealed is not currently used, drilling a replacement well may not be required. In these cases, total costs will be reduced substantially.

D5.3 Additional Logging and Inventory

Given the remedial action of continued monitoring recommended for the site area by this study, contaminated ground water in the surficial aquifers (Drift and Platteville) may continue to migrate to the southeast in response to the regional hydraulic gradient.

TABLE D5-1

COSTS FOR PERMANENT ABANDONMENT/RECONSTRUCTION/REPLACEMENT OF MULTI-AQUIFER WELLS

	Clean Out Cost[a] (Drilling & Crew)	Sealing Cost[a] (Drilling & Crew)	Grout Cost[b]	Replacement Well Cost[c]	Total Abandonment Costs		Total Reconstruction Cost[d]
					With No Replacement Well	With Replacement Well	
Deep Well (1000 ft deep;) 6-10 in. diameter)	\$ 5,000-15,000	\$2,500-4,000	\$6,500-8,000	\$10,000-20,000	\$15,000-30,000	\$25,000-50,000	\$10,000-25,000
Shallow Well (250 ft deep; 6-10 in. diameter)	2,000-5,000	1,000-1,500	2,000-2,500	10,000 20,000	5,000-10,000	15,000 30,000	3,000-7,000

[a] Based on \$800/day for drilling and crew (\$100/hour).
[b] Based on \$8/cubic yard for grout.
[c] Based on estimate of \$36-\$48/foot for 6-inch to 10-inch well pipe (Layne-Western 1983)
[d] Deep wells will be replaced by shallow wells.

Ground-water modeling predicts that the contaminants will migrate to the south and east at a maximum rate of approximately 0.5 feet per day. (Appendix E). Attenuation of contaminants due to adsorption and/or biodegradation should result in very slow migration, however. It is important to identify and study multi-aquifer wells connecting present or likely future areas of contamination in the Drift-Platteville and St. Peter with underlying aquifers. From such a study, the migration of contaminants via multi-aquifer wells in this area can be assessed and, if necessary, controlled.

Movement of contaminants in the Prairie du Chien-Jordan aquifer should be monitored, municipal supply wells should be replaced with new Mt. Simon-Hinckley wells as needed, and a contingency plan to use contaminated municipal supply wells as gradient control wells should be prepared (see Chapters 6 and 7). Multi-aquifer wells that connect contaminated areas (current or predicted) of the Prairie du Chien-Jordan with the Mt. Simon-Hinckley also need to be assessed and, if necessary, controlled.

Preparation for assessing and, if necessary, controlling multi-aquifers wells that could be affected by the migration of contaminants in the Drift-Platteville, St. Peter and Prairie du Chien-Jordan aquifers should be initiated in advance of the actual appearance of contaminants. The multi-aquifer well inventory recently completed by Hickok should be adequate for this purpose for multi-aquifer wells that connect the Drift-Platteville and/or St. Peter to lower aquifers, based on the current extent of contamination and the predicted contaminant migration rates in these aquifers. However, the boundaries of a multi-aquifer well survey for wells connecting the Mt. Simon-Hinckley to other aquifers will need to be extended. Additional multi-aquifers well surveys covering further areas for both shallow aquifers (wells that connect the Drift-Platteville and/or St. Peter to lower aquifers) and the Prairie du Chien-Jordan (wells that connect this aquifer to the Mt. Simon-Hinckley) may be required during the next 100 years, if contaminants migrate in these aquifers.

On the basis of a review of the well inventory recently completed by Hickok, it may be determined that further study of existing wells in the inventory area is warranted. These studies would augment the

Hickok inventory by filling information gaps. Wells of unknown depth should be sounded to determine total depth. If sounding results are inconclusive, geophysical, downhole television, and/or flow meter (spinner) logging should be conducted to identify those multi-aquifer wells which allow interaquifer flow. Accurate location and surface elevation measurements should be surveyed for all wells. The estimated cost of these additional studies is from $25,000 to $50,000.

Future multi-aquifer well inventories for expanded areas of the Drift-Platteville and St. Peter should be conducted in the following manner. First, a study area should be defined that includes areas likely to be impacted by contaminant migration in the Drift-Platteville and St. Peter over the following few decades. Second, a file search of U.S. Geological Survey, Minnesota Geological Survey, Minnesota Department of Health, municipal water boards, and driller's records should be conducted. Third, a door-to-door survey of additional wells in the study area should be completed. Finally, a priority list of known or suspected multi-aquifer wells should be assembled for further study. These studies might include water quality analysis, geophysical logging, downhole television logging, or downhole flow meter logging. The estimated cost to conduct an additional survey as described above is from $25,000 to $100,000, depending on the area covered and the number of multi-aquifer wells discovered.

An inventory of multi-aquifer wells that connect to the Mt. Simon-Hinckley should generally follow the procedures outlined above. However, door-to-door surveys should be limited to industrial or commercial well owners, since it is highly unlikely that private homes will have wells extending as deep as the Mt. Simon-Hinckley. The costs of such a survey is estimated to be $20,000 to $50,000. This range is somewhat lower than that for shallow wells surveys, since fewer wells will be involved.

D5.4 Recommended Remedial Actions for Off-Site Multi-aquifer Wells

The basis for ERT's recommended remedial action for off-site multi-aquifer wells is that sealing or reconstructing all identified multi-aquifer wells is not necessary to provide a cost-effective solution to the stated objectives of allowing for present and reasonably foreseeable future uses of ground-water resources, and of providing safe and adequate public water supplies. This is because multi-aquifer wells simply change the patterns and rates of contaminant migration, and the overall ground-water management recommendation by this study (as discussed in Chapters 6 and 7) is to continue to allow contaminants to migrate, with continued monitoring of contaminant migration, replacement of contaminated municipal supply wells in the Prairie du Chien-Jordan, and a contingency plan for instituting gradient control wells, if future migration patterns prove this to be necessary. This is not to say, however, that no multi-aquifer wells should be sealed or reconstructed. Instead, this overall ground-water management solution defines the context under which selected multi-aquifer wells should be assessed and, if necessary, controlled.

Table D5-2 summarizes the recommendations for controlling off-site multi-aquifer wells and the estimated costs of the various actions. Wide ranges are presented for many of the costs because of considerable uncertainty in predicting how many multi-aquifer wells will need to be controlled. Five specific actions are recommended, with a sixth action being dependent on the results of further study. Each of these recommended actions are discussed below.

Multi-aquifer Well Inventory and Logging

Two specific recommendations address further inventorying and logging of multi-aquifer wells. First, information gaps on the characteristics of multi-aquifer wells identified by Hickok's recent survey should be addressed. The results of this study will provide adequate inventory of shallow multi-aquifer wells (those which extend down from the Drift-Platteville and/or St. Peter) for the next 25 to

TABLE D5-2

ESTIMATED COSTS OF RECOMMENDED

REMEDIAL ACTIONS FOR OFF-SITE

MULTI-AQUIFER WELLS

Recommended Remedial Action	Current Expenditures	Fixed Expenditures (made every 25 to 50 years)
1. Fill information gaps in Hickok's recent multi-aquifer well survey	$25,000 - $50,000	Not Applicable
2. Repeat multi-aquifer well inventory for shallow aquifer as contaminants migrate	Not Applicable	$25,000 - $100,000
3. Expand inventory of multi-aquifer wells connecting the Prairie du Chien-Jordan to the Mt. Simon-Hinckley	$20,000 - $50,000	$20,000 - $50,000
4. Seal or reconstruct multi-aquifer wells which connect:		
a. Shallow aquifers to the Mt. Simon-Hinckley	$0 - $300,000[a]	$0 - $300,000[a]
b. Prairie du Chien-Jordan to the Mt. Simon-Hinckley	$0 - $150,000[b]	$0 - $150,000[b]
5. Assess the significance of multi-aquifer wells identified in item 6	$25,000[e]	$25,000[e]
6. If shown to be cost-effective as a result of Item 5 - seal or reconstruct multi-aquifer wells which connect:		
a. Drift-Platteville to the St. Peter	$0 - $200,000[c]	$0 - $200,000[c]
b. Drift-Platteville and/or St. Peter to the Prairie du Chien-Jordan	$0 - $300,000[d]	$0 - $300,000[d]

TABLE D5-2 (Continued)

Recommended Remedial Action	Current Expenditures	Fixed Expenditures (made every 25 to 50 years)		Total Possible Expenditure Range
		Every 25 Years	Every 50 Years	
Total Present Value Cost				
a. Without Item 6	$70,000 - $575,000	$30,000 - $260,000	$6,000 - $60,000	$76,000 - 835,000
b. With Item 6	$70,000 - $1,075,000	$30,000 - $470,000	$6,000 - $105,000	$76,000 - 1,545,000
Total present value cost for shallow wells (Items 1,2,4a and 6a)				
a. Without Item 6a	$25,000 - 350,000	$10,000 - 165,000	$2,000 - 40,000	27,000 - 515,000
b. With Item 6a	$25,000 - 550,000	$10,000 - 250,000	$2,000 - 60,000	27,000 - 800,000
Total present value cost for deep wells (Items 3, 4b,5 and 6b)				
a. Without Item 6b	$45,000 - 225,000	20,000 - 95,000	4,000 - 20,000	49,000 - 320,000
b. With Item 6b	$45,000 - 525,000	20,000 - 225,000	4,000 - 50,000	49,000 - 750,000

Notes to Table D5-2.

(a) Based on zero to ten wells being sealed at a cost of $15,000 to $30,000 per well (see (Table D5-1).

(b) Based on zero to five wells being sealed at a cost of $15,000 to $30,000 per well (see Table D5-1).

(c) Based on zero to twenty wells being sealed at a cost of $5,000 to $10,000 per well (see Table D5-1).

(d) Based on zero to ten wells sealed at a cost of $15,000 to $10,000 per well (see Table D5-1).

(e) Study costs for evaluating ground-water monitoring data and for performingl simplified ground-water modeling.

50 years, based on predicted migration rates of contaminants in these aquifer (see Chapter 6 and Appendix E). This inventory of shallow multi-aquifer wells should be expanded to wider areas as contaminants migrate. This should be done every 25 to 50 years, depending on how fast contaminants migrate.

The second recommendation is that the inventory of multi-aquifer wells that connect the Prairie du Chien-Jordan and Mt. Simon-Hinckley should be extended now to cover a wider area, and should be extended further as needed on a regular basis (every 25 to 50 years) if contaminants in the Prairie du Chien-Jordan continue to migrate.

Multi-aquifer Well Reconstruction or Sealing

Because the Mt. Simon-Hinckley may be used as an alternate source of public water supply, it is important that multi-aquifer wells which could transmit contaminants into this aquifer be sealed or reconstructed. It should also be noted, however, that there is some margin for error in this effort, since transport times in the Mt. Simon-Hinckley are long, and significant dilution, dispersion and attenuation of any undetected contaminants which enter the aquifer should be substantial.

It is also recommended that further limited study be performed to better assess the significance of multi-aquifer wells that connect the Drift-Platteville, St. Peter and/or Prairie du Chien-Jordan aquifers. It is not apparent from the results of this study that sealing or reconstructing these types of multi-aquifer wells would be cost-effective in reducing the rate at which the contaminated area of the Prairie du Chien-Jordan or St. Peter aquifers expands or in increasing the rate at which these aquifers are effectively purged of contaminants by pumping, adsorption or dispersion. A limited study that incorporates the results of Hickok's recent multi-aquifer well survey with ground-water modeling aimed at predicting contaminant migration is required to better resolve this cost-effectiveness question. If the results of this study indicate that sealing or reconstructing multi-aquifer wells that connect the Drift-Platteville, St. Peter and/or Prairie du Chien-Jordan aquifers is indeed cost-effective in helping to achieve the stated objectives of this study, then this should be done. Table D5-2 includes estimated costs for sealing or reconstructing these types of multi-aquifer wells.

Estimated Costs

Table D5-2 shows that the present values cost of the five recommended remedial actions for off-site multi-aquifer wells ranges from $76,000 to $835,000. This wide range is largely due to uncertainties in the number of multi-aquifer wells that will need to be sealed or reconstructed. It also reflects uncertainty as to how frequently multi-aquifer well inventories will need to be expanded. The actual cost is likely to be less than half a million dollars, assuming that the number of Mt. Simon-Hinckley multi-aquifer wells requiring remedy is less than the worst-case projection of 15 wells every 25 years. This seems likely because few such wells have been identified to date, and finding a great many more is unlikely, in light of the intensive study of the industrial/commercial area near the site and the fact Mt. Simon-Hinckley wells are only expected to occur in such areas.

Table D5-2 shows that the estimated present value cost to remedy multi-aquifer wells connecting the Drift-Platteville, St. Peter or Prairie du Chien-Jordan could be as high as $750,000 in a worst-case projection (30 wells requiring remedy every 25 years). Again, if it is shown that remedying such wells is indeed cost-effective, the actual cost is likely to be less than half a million dollars.

It should be noted that the cost estimates given in Table D5-2 do not allow for new wells to replace multi-aquifer wells that are sealed. This reflects the assumption that every reasonable effort would be made to avoid this additional expense by reconstructing multi-aquifer wells that are in use or to minimize this additional expense by drilling shallow (Drift-Platteville or St. Peter) wells.

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APPENDIX E

GROUND-WATER MODELING

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E1. INTRODUCTION

E1.1 Objectives

A ground-water model of the St. Louis Park area was developed to examine the transport of contaminants and to evaluate alternative programs for the control of contaminant movement in the ground-water system. Movement of contaminants in ground water is largely determined by the hydraulics of the ground-water system: that is, the flow of water determines the movement of any dissolved contaminants. To a lesser degree, the transport of contaminants is also influenced by contaminant dispersion and chemical or physical reactions which remove or immobilize contaminants.

The goals for the modeling study were as follows:

- predict the direction and degree of contaminant transport from sources of contamination;
- assess the importance of transport between aquifers as a cause of contamination; and,
- evaluate the ability of alternative schemes to control the movement of contaminants.

Ultimately, the findings of the modeling study are used in the preparation of recommendations for a preferred control program.

E1.2 Hydrogeology and Physical Setting

A thorough understanding of the hydrogeology of the St. Louis Park area is a prerequisite to construction of the ground-water model. This section briefly describes the hydrogeology of the modeling area, with an emphasis on the defintion of hydrogeologic units to be used in the model. A more detailed description of the area's geology may be found in Norvitch et al. (1973), which served as the primary reference for this section.

The Minneapolis - St. Paul area lies on a geologic structure commonly referred to as the Twin Cities artesian basin. Consolidated sedimentary rocks of Precambrian, Cambrian, and Ordovician age were deposited in a north-south trending trough on the Precambrian rock surface. As revealed in the geologic column, no other deposition occurred in this area until Quaternary time. A generalized geologic column of the study area is shown in Table E1-1.

During the Pleistocene Epoch, the erosional bedrock surface was covered with glacial drift. The drift is in contact with formations as far down the geologic section as the St. Lawrence Formation due to the fact that the bedrock surface was dissected by many valleys and channels. These drift-filled channels are important hydrologically because they provide hydraulic continuity between deep bedrock aquifers and surficial deposits, and hydraulic connection between deep bedrock aquifers and major rivers in the area (Guswa et al. 1982).

General water-bearing characteristics of the geologic units are also given in Table E1-1. Ground water may be obtained from eight of the geologic units although four of these units comprise the two primary aquifers in the area. The Prairie du Chien Group (predominantly a highly-fractured dolomite) and the Jordan Sandstone together supply about 75 per cent of ground water pumped in the Minneapolis-St. Paul area. Another 15 per cent of total ground-water supplies is obtained from the Mt. Simon and Hinckley Sandstones, which constitute the deepest aquifer unit. Lesser quantities of ground water can be pumped from the glacial drift, Platteville Limestone, St. Peter Sandstone, and Ironton and Galesville Sandstones.

Based upon the water-bearing characteristics summarized in Table E1-1, nine hydrogeologic units are defined for the study area. These units are shown on the same table in relation to the geologic units.

The Mississippi and Minnesota Rivers are along the eastern and southern edge of the study area, respectively. As will be described in Section E2.1, these rivers are used as two of the boundaries for the flow model.

A continental-type climate is predominant in the study area with an average annual precipitation of 28.3 inches and average annual evapotranspiration of 22.5 inches (Norvitch et al. 1973).

TABLE E1-1
GEOLOGIC UNITS AND THEIR WATER-BEARING CHARACTERISTICS
(After Norvitch et al. 1973)

Hydrogeologic Units Defined for this Study	System	Geologic Unit	Approx. Range in thickness (feet)	Description	Water-Bearing Characteristics
Drift-Platteville Aquifer	Quarternary	Undifferentiated Glacial Drift	0-400+	Glacial till, outwash sand and gravel, lake deposits, and alluvium.	Distribution of aquifers and relatively impermeable confining beds is poorly known, especially in subsurface.
	Ordovician	Decorah Shale	0-95	Shale, bluish-green to bluish-gray; blocky.	Confining bed.
		Platteville Limestone	0-35	Dolomitic limestone and dolomite, dark-gray, hard.	Water is generally under artesian pressure where overlain by Decorah Shale. Not considered to be an important source of water in area of study.
Glenwood Confining Bed		Glenwood Shale	0-18	Shale, bluish-gray to bluish-green; generally soft.	Confining bed.
St. Peter Aquifer Basal St. Peter Confining Bed		St. Peter Sandstone	0-150+	Sandstone, white, fine to medium-grained, well-sorted. 5-50 feet of siltstone and shale near bottom of formation.	Water occurs under both confined and unconfined conditions. Confining bed near bottom of formation hydraulically separates sandstone from underlying Prairie du Chien-Jordan aquifer. Suitable source for domestic supplies.
Prairie du Chien-Jordan Aquifer		Prairie du Chien Group Shakopee Dolomite	0-250+	Dolomite, light-browm to buff.	Together, the Prairie du Chien Group and the Jordan Sandstone constitute the major aquifer unit. The two are hydraulically connected throughout most of the area
		New Richmond Sandstone	0-250+	Sandstone and sandy domomite, often missing.	
		Oneota Dolomite	0-250+	Dolomite, light-brownish-gray to buff.	
	Cambrian	Jordan Sandstone	0-100+	Sandstone, white to yellowish.	
		St. Lawrence Formation	0-65	Dolomitic siltstone and fine-grained dolomitic sandstone.	Confining bed.
St. Lawrence-Franconia Confining Bed		Franconia Formation	0-200+	Sandstone, very fine grained. Some interbedded shale and glauconitic sandstone.	Not considered to be an important water source in the area.
Ironton-Galesville Aquifer	Cambrian	Ironton Sandstone	0-80+	Sandstone, white, medium- to fine-grained, silty.	An important aquifer beyond the limits of the Prairie du Chien-Jordan aquifer.
		Galesville Sandstone	0-80+	Sandstone, yellow to white, medium- to coarse-grained.	

TABLE E1-1 (Continued)

Hydrogeologic Units Defined for this Study	System	Geologic Unit	Approx. Range in thickness (feet)	Description	Water-Bearing Characteristics
Eau Claire Confining Bed		Eau Claire Sandstone	0-150	Sandstone, siltstone, and shale, gray to reddish-brown.	Confining Bed.
Mt. Simon-Hinckley Aquifer		Mt. Simon Sandstone	As much as 200	Sandstone, gray to pink, medium- to coarse-grained.	Secondary major aquifer, supplies about 15 percent of ground water pumped in the metropolitan area.
	Precambrian	Hinckley Sandstone	As much as 200	Sandstone, buff to red, medium- to coarse-grained.	
Impermeable Bedrock		Red Clastics	As much as 4000	Silty feldspathic sandstone and lithic sandstone.	Data are lacking in metropolitan area.
		Volcanic rocks	As much as 20,000	Mostly mafic lava flows.	Deeply buried in metropolitan area.

E2. MODEL DEVELOPMENT

E2.1 Model Construction

The primary tool used in this study is the three-dimensional aquifer simulation model developed by the United States Geological Survey (Trescott 1975 and Trescott and Larson 1976). The U.S Geological Survey (USGS) model simulates the flow of ground water and changes in ground-water storage for a three-dimensional aquifer system. The model is capable of simulating either time constant (steady state) or time variable (transient) flow and storage conditions.

Simulation of ground-water flow through time was not performed in this study due to limitations posed by data inadequacies and practical considerations. The period of interest is long -- over sixty years -- and adequate data do not exist to recreate historical conditions for a comprehensive simulation. As well, pumping operations are highly irregular: a particular pump will be operated only part of the time, often only a few per cent of the time. While operating, the well is pumped at full capacity, thus exerting a significant influence on the nearby aquifer during the pumping period. This influence ceases when pumping stops. The consequent pattern is one of sporadic, transient flow patterns in response to the current nearby pumping wells. The complexity of this pattern through time precludes retrospective or predictive simulations in other than an approximate long-term average sense.

The USGS model also includes an option for quasi-three-dimensional systems. With this option, the ground-water system is modeled as a vertical series of layered aquifers. The flow within any single aquifer is modeled as horizontally two-dimensional, with vertical flow between adjacent layers. The individual layers may be confined, leaky or free-surface aquifers. The characteristics of confining beds separating aquifer layers are captured via inter-layer leakage coefficients. The quasi-three-dimensional model, if suited to the ground-water system to be modeled, is a cost-effective simplification of the hydrogeology. It is highly appropriate for the multiple aquifer system of the St. Louis Park area.

A ground-water flow model, as opposed to a ground-water transport model, has been selected as the study's primary modeling tool for a variety of reasons. In particular, a flow model is consistent with both the purposes of the modeling study and the data base available to support model construction and verification. As stated above, ground-water flow determines contaminant transport to first approximation. Thus, a ground-water flow model can supply a good indication of the rate and direction of contaminant migration. The flow model requires fewer parameters and thus a less extensive data base than transport models. The data needs for transport models are particularly onerous since they require parameters which are very difficult to measure and seldom available. This is an important consideration since a model based on uncertain or sparse data is subject to questions of reliability and accuracy.

E2.1.1 Vertical Structure

The quasi-three-dimensional approach of the model is used in this study to construct a model of the St. Louis Park area ground-water system. The general character of the model's vertical structure is indicated in Figure E2-1. This generalized structure may not necessarily be locally accurate throughout the entire model since certain of the layers do not exist everywhere within the model boundaries. The constructed model considers the difference in aquifer thickness and vertical elevation as indicated by available data, thus accounting for local variablity.

Included in the model are five major aquifers underlying the St. Louis Park area. Proceeding from the lowermost, there are four confined aquifer layers: the Mt. Simon-Hinckley (layer 1), the Ironton-Galesville (layer 2), the Prairie du Chien-Jordan (layer 3), and the St. Peter Sandstone (layer 4). These are overlain by an unconfined aquifer (layer 5), which combines the Platteville Limestone and Basal, Middle and Upper Drift formations. Although a number of the aquifer layers include more than one geologic formation, each layer is well approximated as a single hydrogeologic unit.

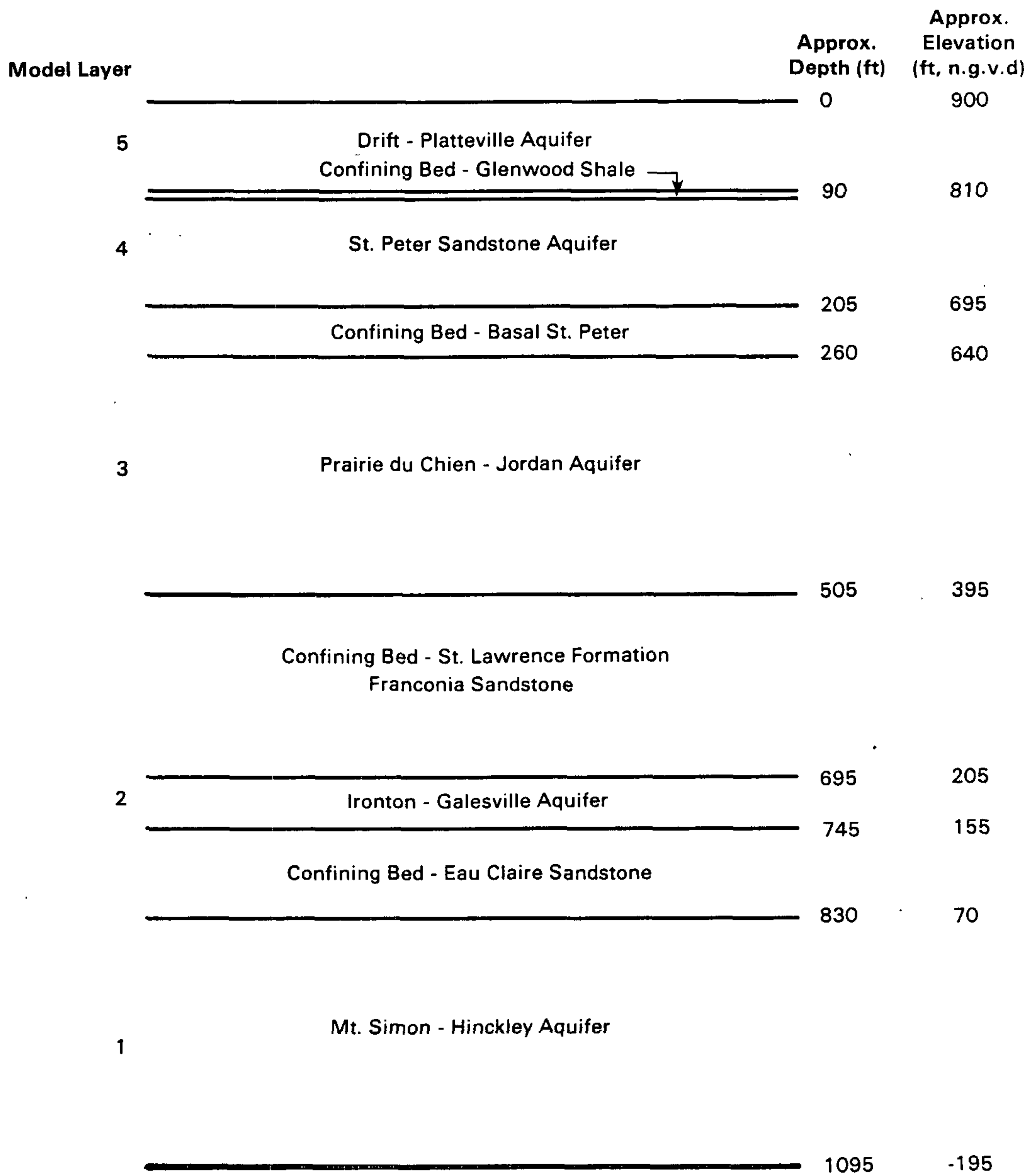

Figure E2-1 Vertical Aquifer Geometry in Model

Figure E2-2 is an approximate depiction of the layered model structure, showing the four bedrock aquifers, the valley dissections, and the overlying glacial drift.

E2.1.2 Horizontal Structure

The horizontal boundaries of the model are placed so as to correspond with natural physical boundaries of the ground-water system. The finite difference grid and enclosing boundaries are shown in Figure E2-3. The overall dimension of the modeled area is 22 miles east-west and 16 miles north-south. The southern and eastern boundaries of the model correspond approximately with the Minnesota and Mississippi Rivers, into which the Prairie du Chien and overlying aquifers discharge. To the north and west, the model boundaries correspond roughly with ground-water flow divides as indicated in potentiometric surface contour maps in Norvitch et al. (1973).

As seen in Figure E2-3, a variable size finite difference grid has been constructed for this study. At the center of the study area, at St. Louis Park, the grid spacing is smallest: nodes are square, one-half mile by one-half mile. One-half mile was selected as the smallest node size since greater detail is not supported by the available hydrogeologic information, and since one-half mile is in approximate agreement with the size of the zones of capture predicted by Hickok and Associates (1981) for proposed recovery wells.

Away from the center of the model, the grid spacing is gradually increased to 1.5 or 2 mile nodes at the boundaries. Larger grid spacing is employed at the model periphery since ground-water flow and contaminant transport beyond the central area is less critical to the study results. The results in the central area are not highly dependent on the location of the distant boundaries.

E2.1.3 Boundary Conditions

As stated above, the model boundaries are located so as to minimize the influence of the boundaries upon hydraulic predictions in the center of the model. To this end, the boundaries are located

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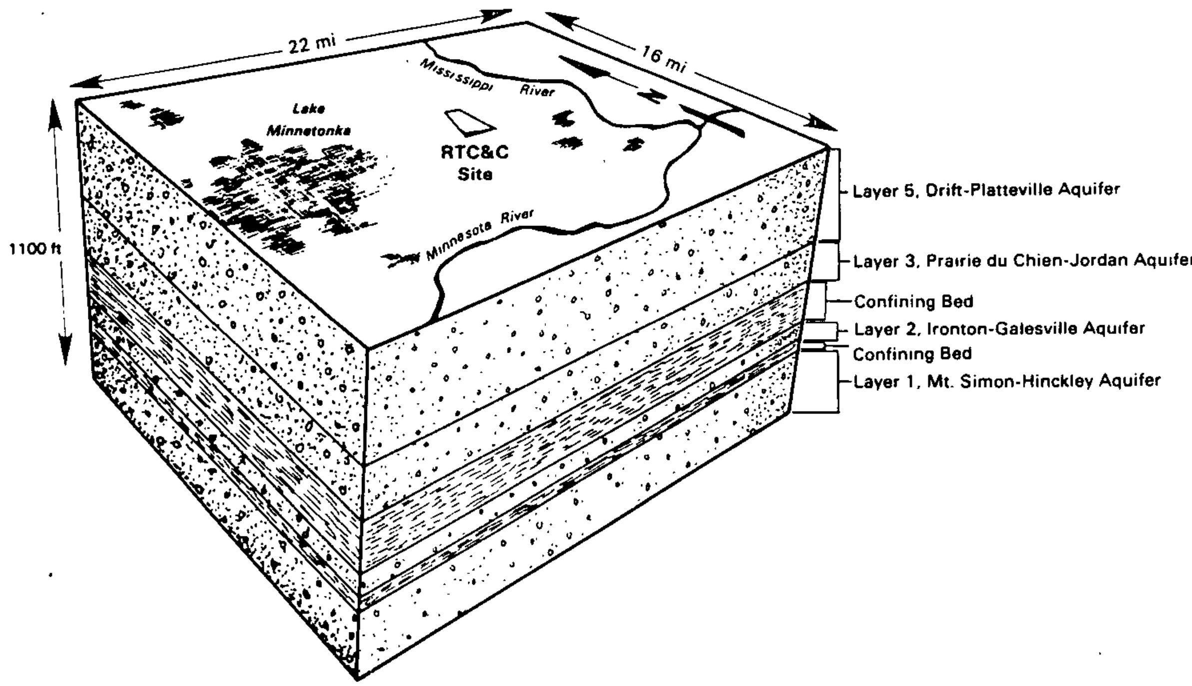

(a) View of Model Showing Surface Features and Drift-Platteville Aquifer

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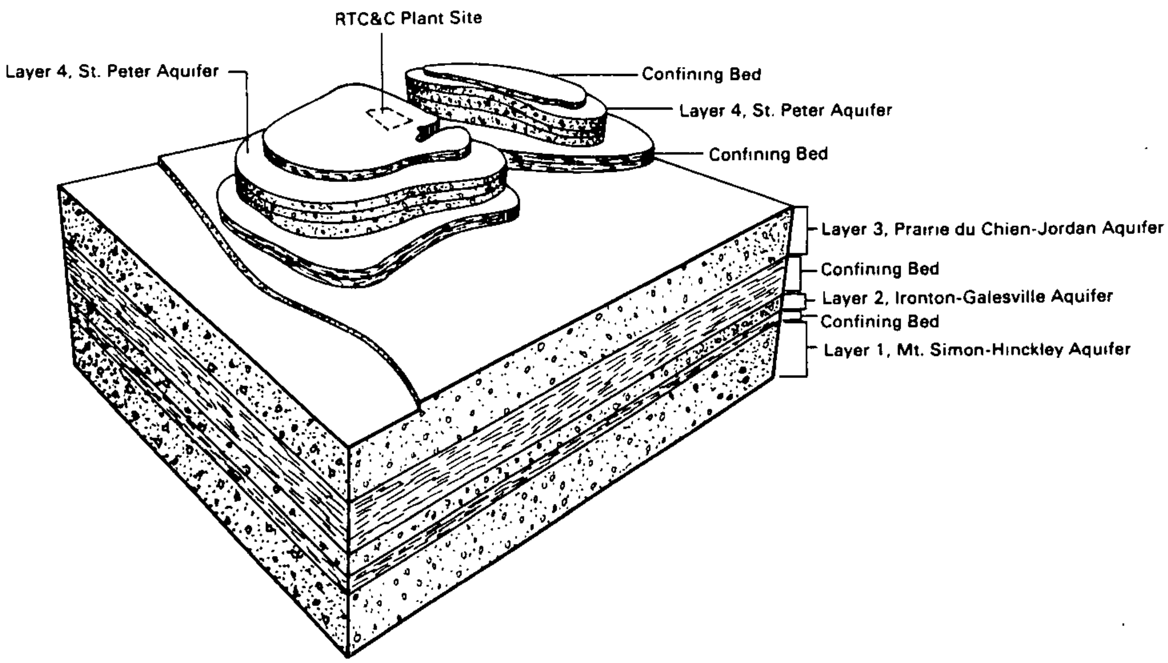

(b) View of Model Layers Beneath Drift-Platteville Aquifer

Figure E2-2 Three-Dimensional Construction of Ground-Water Model

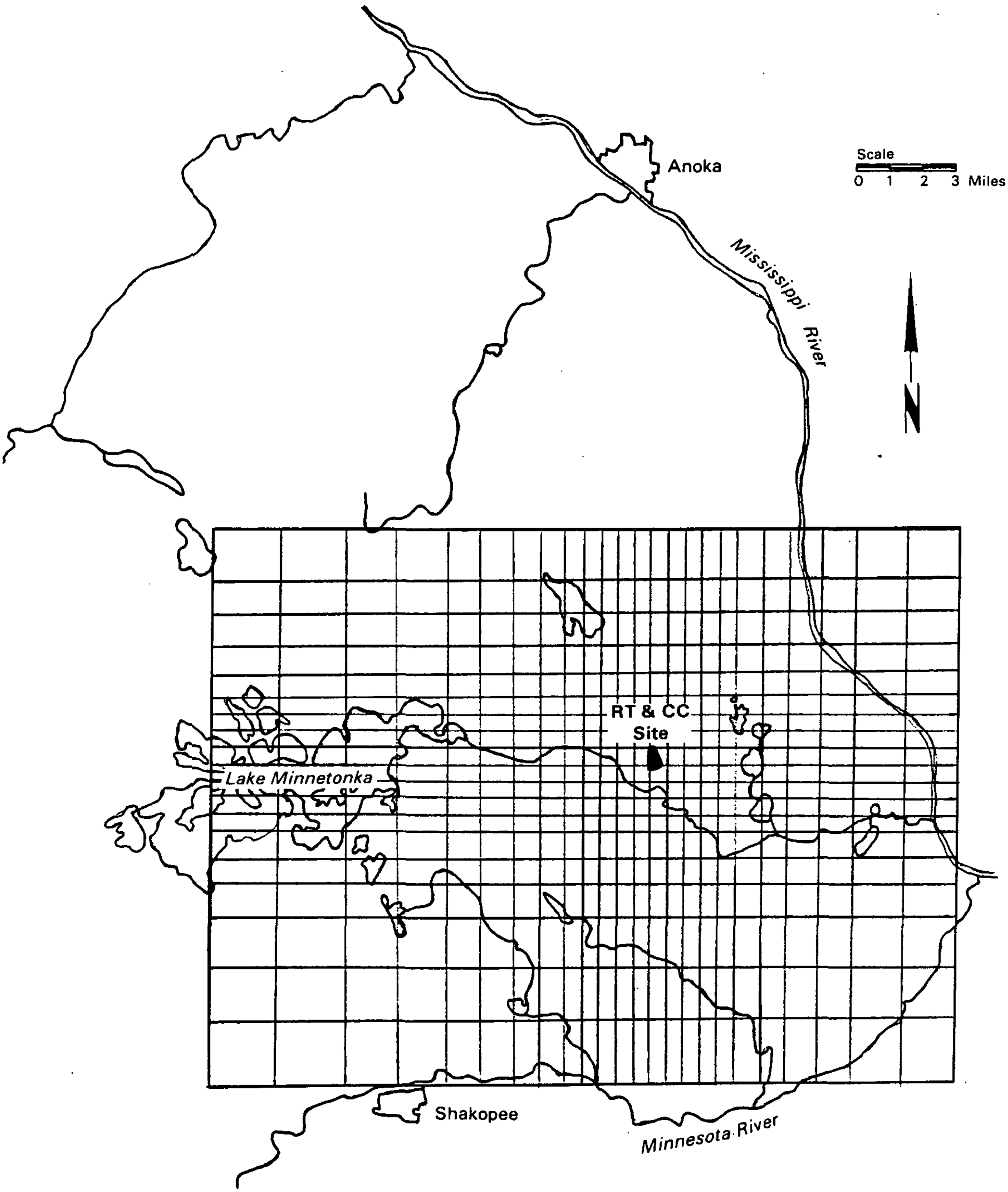

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Figure E2-3 Map of St. Louis Park Area Showing Model Boundaries relatively distant from the area of interest and are placed at natural boundaries of the actual ground-water system. A requirement of the USGS computer program is that the model area be entirely circumscribed by a boundary of no-flow nodes. Physically-based boundary conditions (such as flow or constant head nodes) are then specified within this outermost computational boundary. Constant head boundaries are specified in the model to locate the Mississippi and Minnesota Rivers. Seepage boundaries with a lower hydraulic conductivity were placed along constant head river nodes in the Drift-Platteville aquifer layer. For the most part, flow in the Prairie du Chien-Jordan, St. Peter, and Drift-Platteville aquifers discharges to the rivers. However, near local pumping centers there may be induced recharge from the river. This is consistent with the potentiometric surface contours indicated by Norvitch et al. (1973).

Along the western and northern boundaries, no-flow boundaries are employed for the Prairie du Chien-Jordan and Drift-Platteville aquifers while flow boundaries are specified for the underlying layers. Flux values are determined from the observed hydraulic gradients and estimated aquifer properties in these lower layers. Figures E2-4 through E2-8 show the boundary conditions for each of the five aquifer layers. Major municipal and industrial pumping centers within each aquifer are also depicted on the figures as pumping wells.

Figures E2-4 through E2-8 show the standard node numbering system used in the USGS model. The index I numbers nodes from west to east, and the index J numbers nodes from south to north. A third index, K, also is used in the model: it numbers layers from bottom to top.

Surface recharge from precipitation and lakes as well as flow between aquifer layers must also be established for the model. Preliminary estimates of recharge and interlayer leakage were made from available information. Adjustment made in these preliminary values during model calibration is discussed in Section E2.4.

E2.2 Supporting Data Base

Available hydrogeologic information was used to select values for aquifer characteristics required in the model. No new studies or

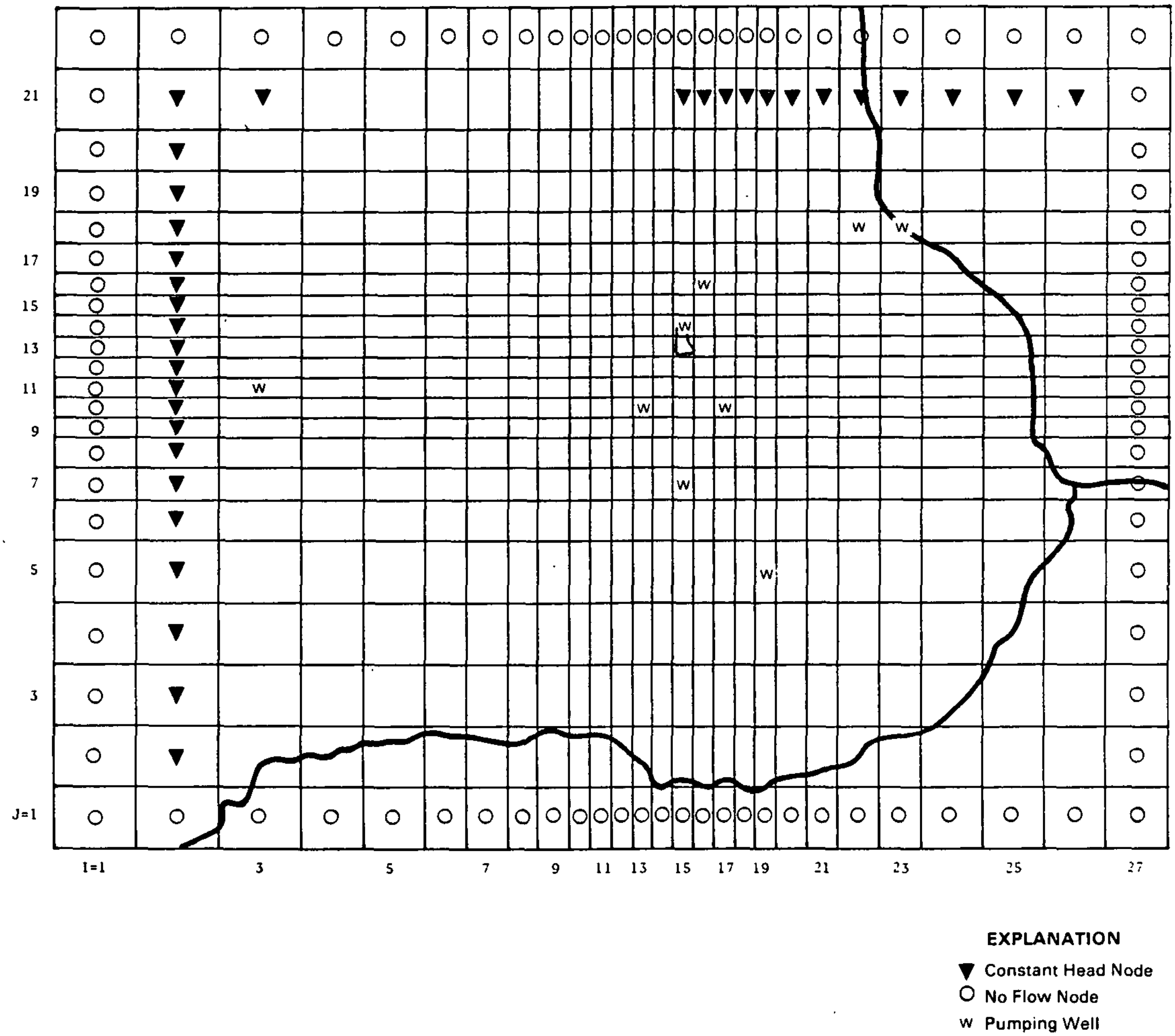

Figure E2-4 Finite Difference Grid for Model Layer 1 (Mt. Simon-Hinckley Aquifer) Showing Boundary Conditions and Pumping Centers

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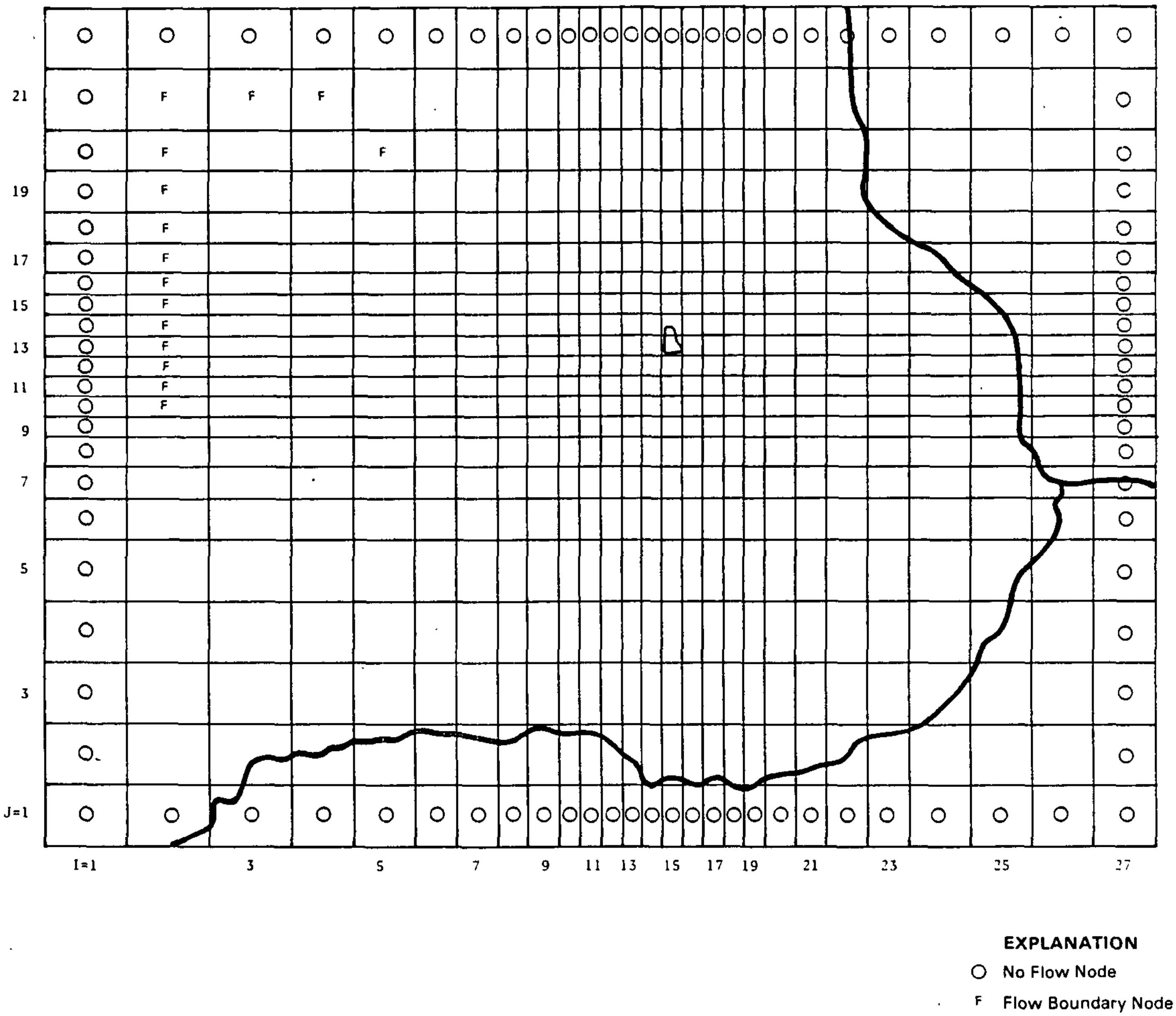

Figure E2-5 Finite Difference Grid for Model Layer 2 (Ironton-Galesville Aquifer) Showing Boundary Conditions and Pumping Centers

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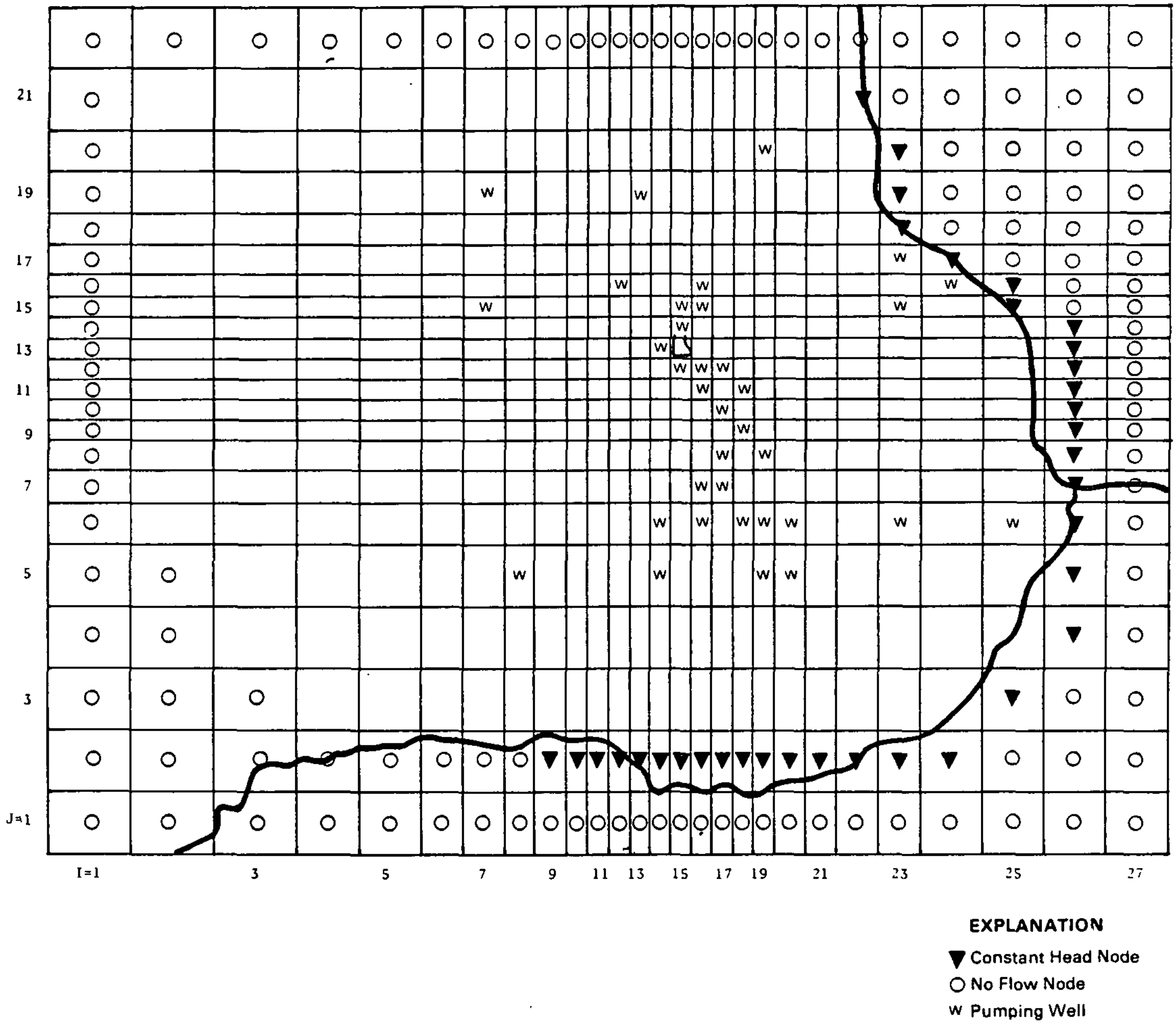

Figure E2-6 Finite Difference Grid for Model Layer 3 (Prairie du Chien-Jordan Aquifer) Showing Boundary Conditions and Pumping Centers

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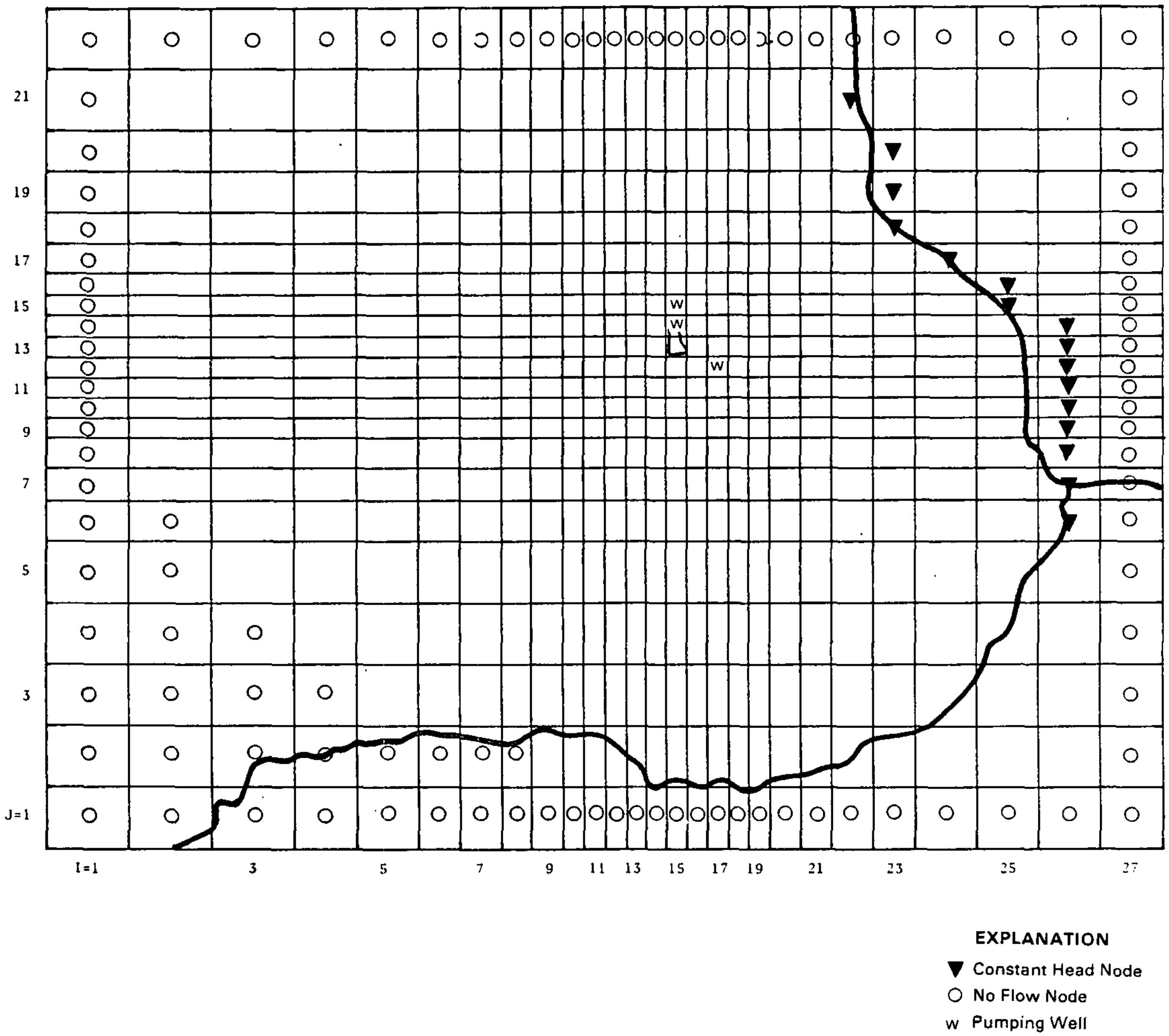

Figure E2-7 Finite Difference Grid for Model Layer 4 (St. Peter Aquifer) Showing Boundary Conditions and Pumping Centers

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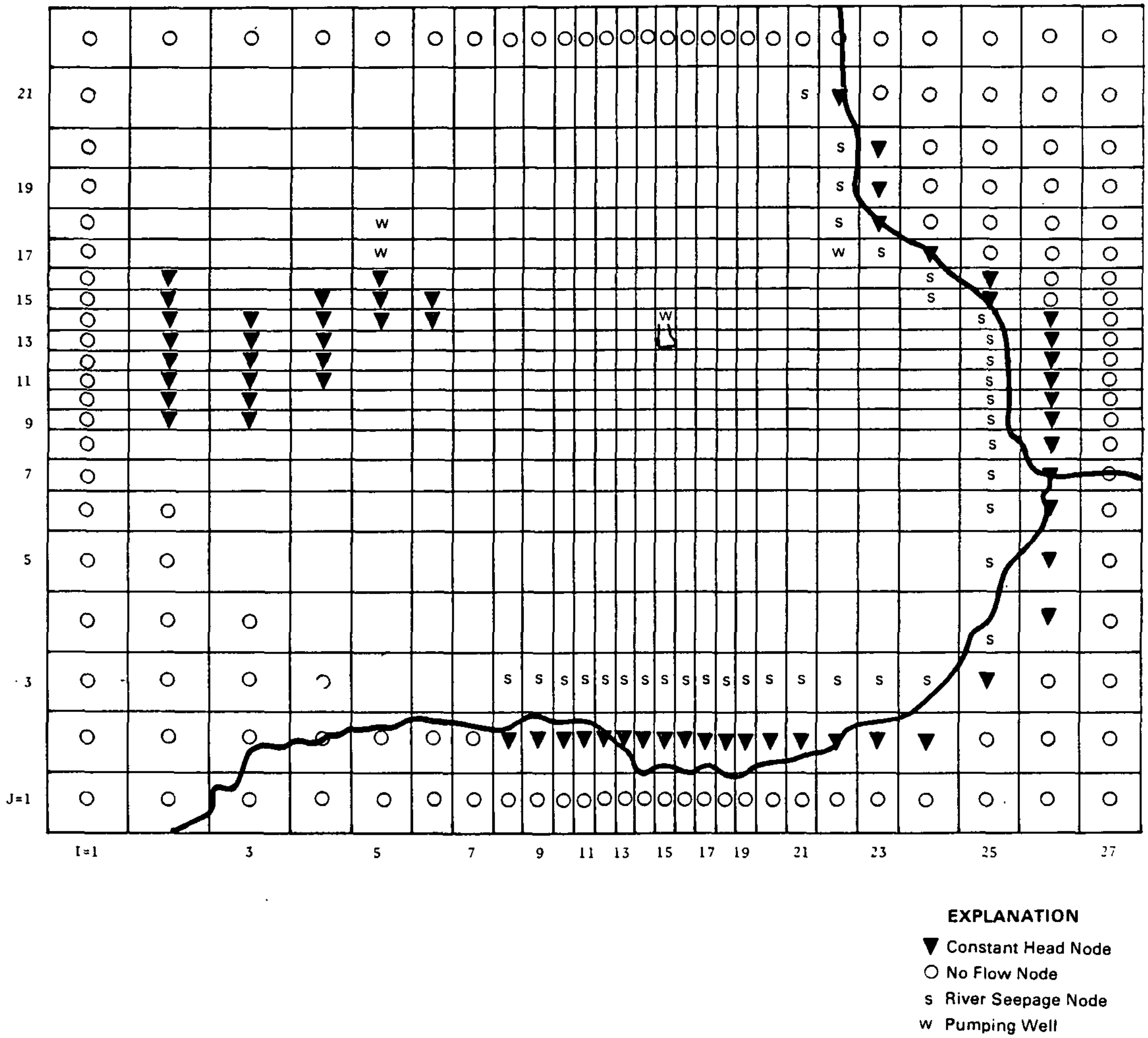

Figure E2-8 Finite Difference Grid for Model Layer 5 (Drift-Platteville Aquifer) Showing Boundary Conditions and Pumping Centers field programs were initiated in order to obtain new data. The input data required for the flow model are listed in Table E2-1.

Initial head distribution information for each aquifer is required to describe ground-water flow direction and gradient. Several USGS publications (Norvitch et al. 1973; Hult and Schoenberg 1981; Larson-Higdem et al. 1975; and Norvitch and Walton 1979) were utilized to describe the initial heads at each node for each of the five aquifers modeled.

Characteristics such as transmissivity (unit rate of flow through an aquifer section), storage coefficient (unit volume of water released from storage during pumping), and hydraulic conductivity (measure of aquifer capacity to transmit water) describe the hydraulic properties of an aquifer. These data are critical to the determination of the rate of ground-water flow and drawdowns associated with pumpage. Values for these characteristics for aquifers in the St. Louis Park area are contained in the publications noted above, as well as Reeder et al. (1976) and Mogg (1962). Where an aquifer consists of more than one geologic formation, differences in formation characteristics were accounted for by using average properties appropriate to the entire aquifer.

Transmissivity values for each confined aquifer in the model are specified in units of square feet per second (ft^2/sec) and gallons per day per foot (gpd/ft) as follows:

	(ft^2/sec)	(gpd/ft)	Reference
Mt. Simon-Hinckley	0.01	6,500	(ERT calibration)
Ironton-Galesville	0.00024	155	(Norvitch et al. 1973)
Prairie du Chien-Jordan	0.0864	56,000	(Reeder et al. 1976)
St. Peter	0.033	21,000	(Mogg, 1962)

Only the Prairie du Chien-Jordan aquifer (layer 3) is modeled as heterogeneous because of variability in aquifer thickness described in Norvitch et al. (1973) and Hult and Schoenberg (1981). Figure E2-9 shows the variability in transmissivity for this layer by node. The low transmissivity values beneath the site are a consequence of a moderate thinning of the Jordan formation in that area; this does not lead to significantly different flow behavior at the site.

TABLE E2-1

INPUT DATA REQUIRED FOR USGS 3-D MODEL

Initial head distribution

Storage coefficient (for transient simulations only)

Transmissivity (for each confined aquifer)

Hydraulic conductivity and bottom elevations (for unconfined aquifers)

Interlayer leakage coefficients

Recharge from precipitation

Location of existing pumping wells and discharge

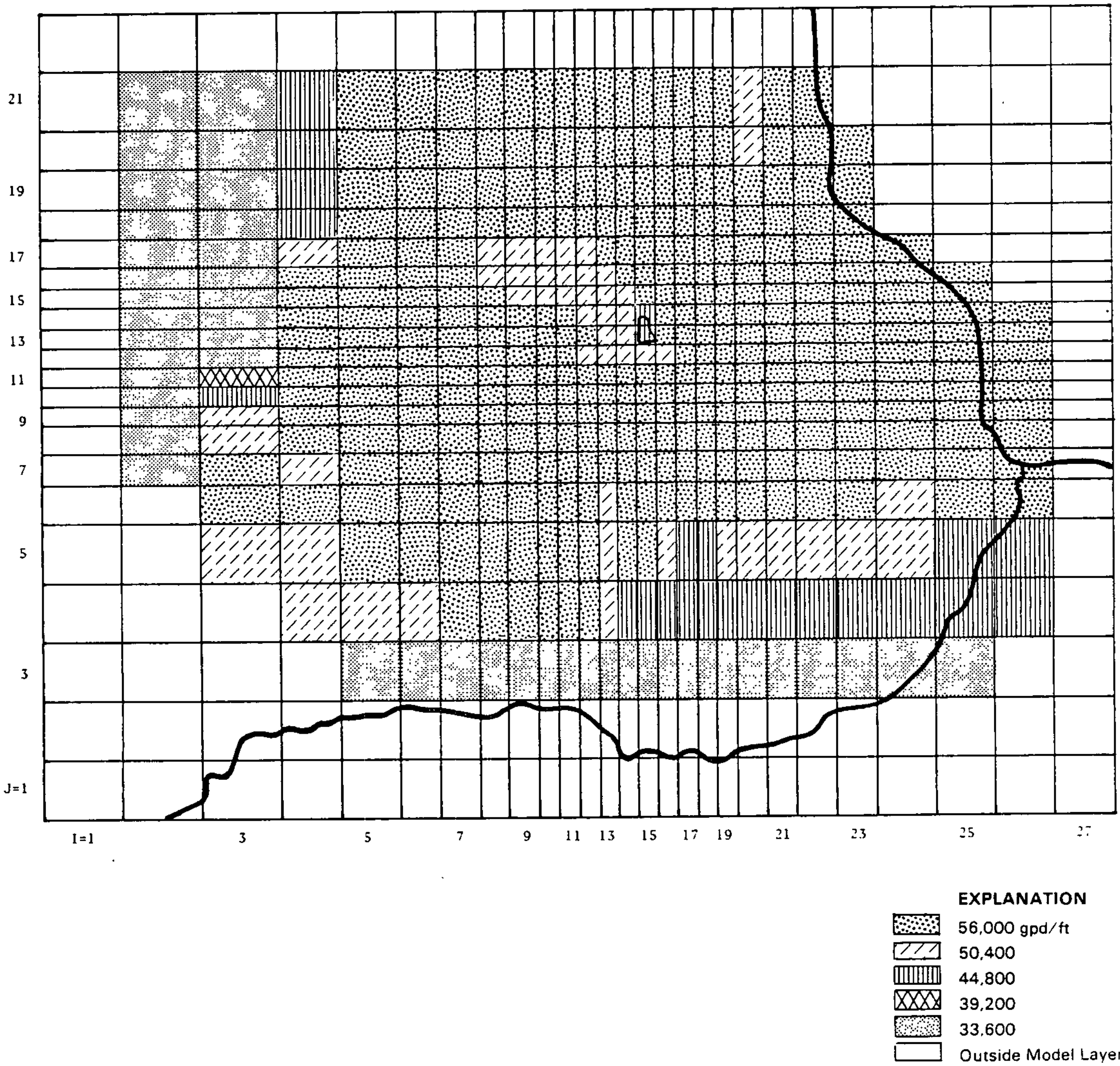

Figure E2-9 Variation of Transmissivity in Model Layer 3 (Prairie du Chien-Jordan Aquifer)

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Hydraulic conductivity and bottom elevation were specified for the Drift-Platteville aquifer in the model since this layer is unconfined. A value for hydraulic conductivity of 7.5×10^{-4} feet per second was used at every node within the layer boundaries except along the Mississippi and Minnesota Rivers where seepage nodes were specified (see Figure E2-8). Horizontal hydraulic conductivity was set at 7.5×10^{-5} feet per second in the seepage nodes. The variation in bottom elevation within this layer which was used in the model is given in Figure E2-10.

No values for storage coefficients were required for the model since only steady state simulations were made.

An accurate model of the ground-water flow regime in the study area must account for interaquifer leakage and recharge. Recharge to the system was based on rainfall data. A value of 7.5 inches per year was used as recharge to the upper unconfined layer (Drift-Platteville aquifer).

Interlayer leakage coefficients are based on the thickness and vertical hydraulic conductivity of the confining layer. Values used in the model to describe interaquifer leakage were obtained primarily from Larson-Higdem et al. (1975) and Barr (1977). During calibration, initial values for the Mt. Simon-Hinckley and Ironton-Galesville aquifers were slightly adjusted. Final values used in the model are as follows (units are per second):

Leakage to:	Mt. Simon-Hinckley (through Eau Claire confining bed)	1.0×10^{-12}
	Ironton-Galesville (through St. Lawrence-Franconia confining bed)	7.0×10^{-11}
	Prairie du Chien-Jordan (through basal St. Peter)	1.25×10^{-10}
	St. Peter (through Glenwood confining bed)	1.6×10^{-12}

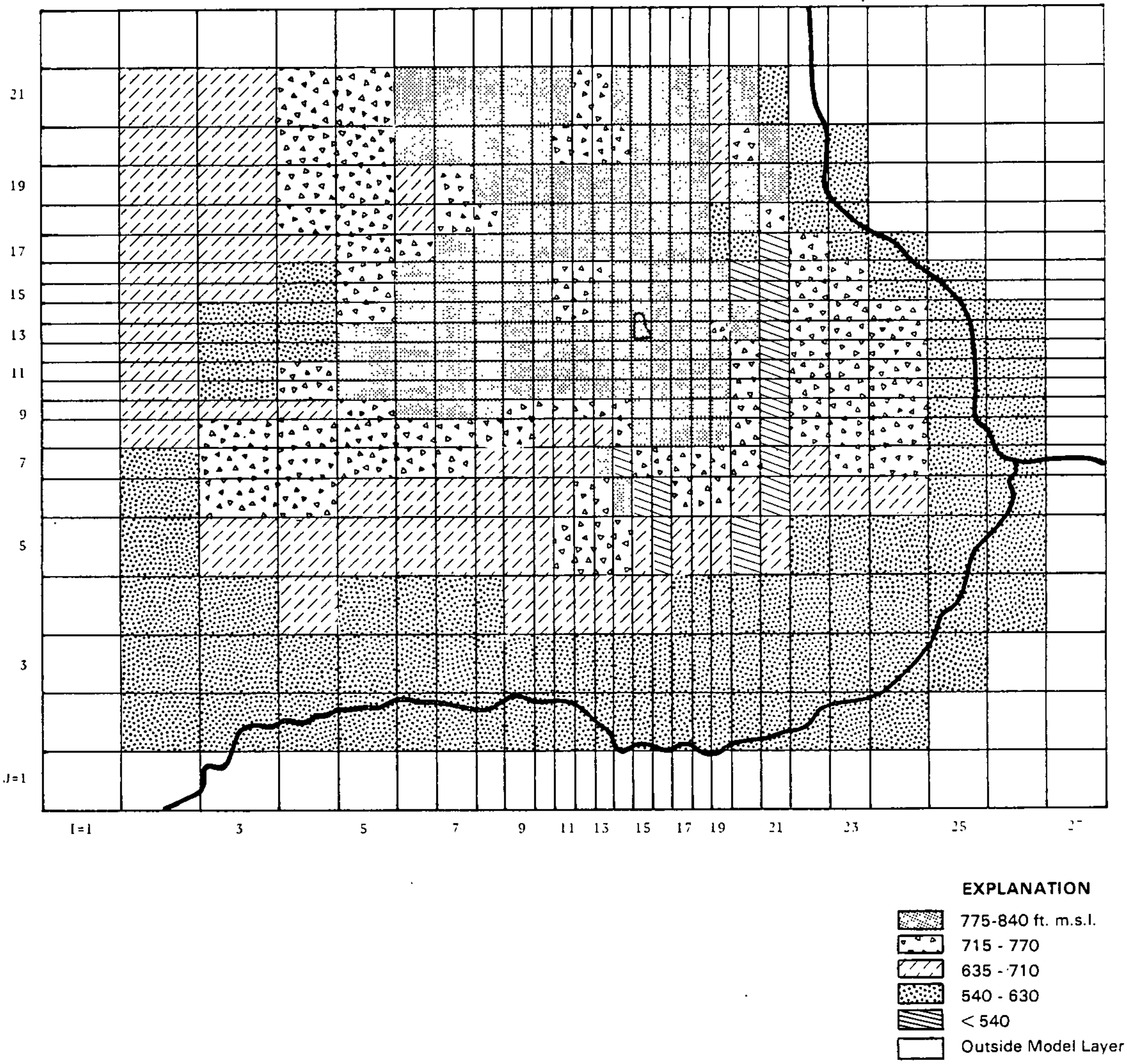

Figure E2-10 Variation in Bottom Elevation in Model Layer 5 (Drift-Platteville Aquifer)

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Variability in leakage between certain layers (Drift-Platteville to St. Peter and St. Peter to Prairie du Chien-Jordan) was required because of the absence of confining layers between these units in some areas. Confining layers are absent due to prehistoric erosion, for example where ancient rivers cut valleys into the bedrock. Where confining layers are absent, the leakage values were increased from 500 to 1000 times, causing the layers to behave as if they were hydraulically connected. Figures E2-11 and E2-12 show where confining beds were modeled as absent over layers 3 (Prairie du Chien-Jordan) and 4 (St. Peter).

The location, construction and withdrawal rates of wells in the model area are necessary to accurately calibrate and verify model results. Data regarding local pumping centers from Hult and Schoenberg (1981), Guswa et al. (1982), and MDNR (1982) were used to calibrate the model. Ground-water users, their location on the finite difference grid, and quantity of water pumped are given in Table E2-2 for each aquifer layer.

Previous study has indicated the importance of multi-aquifer wells to the St. Louis Park ground-water flow system (Hult and Schoenberg 1981 and Hult 1979). Limited data on multi-aquifer wells were available to this study from unpublished USGS information (Hult 1979) and was used to estimate flow between model layers via multi-aquifer wells.

E2.3 Travel Path Modeling

As part of the modeling study, the USGS three-dimensional model was extended to incorporate options for calculating and plotting ground-water velocities, travel times, and paths. One subroutine, VELO, was added to the existing model and a separate plotting program was written.

The purpose of subroutine VELO is to compute the velocity of ground-water flow by solving the following equation (in this case, for the x-direction):

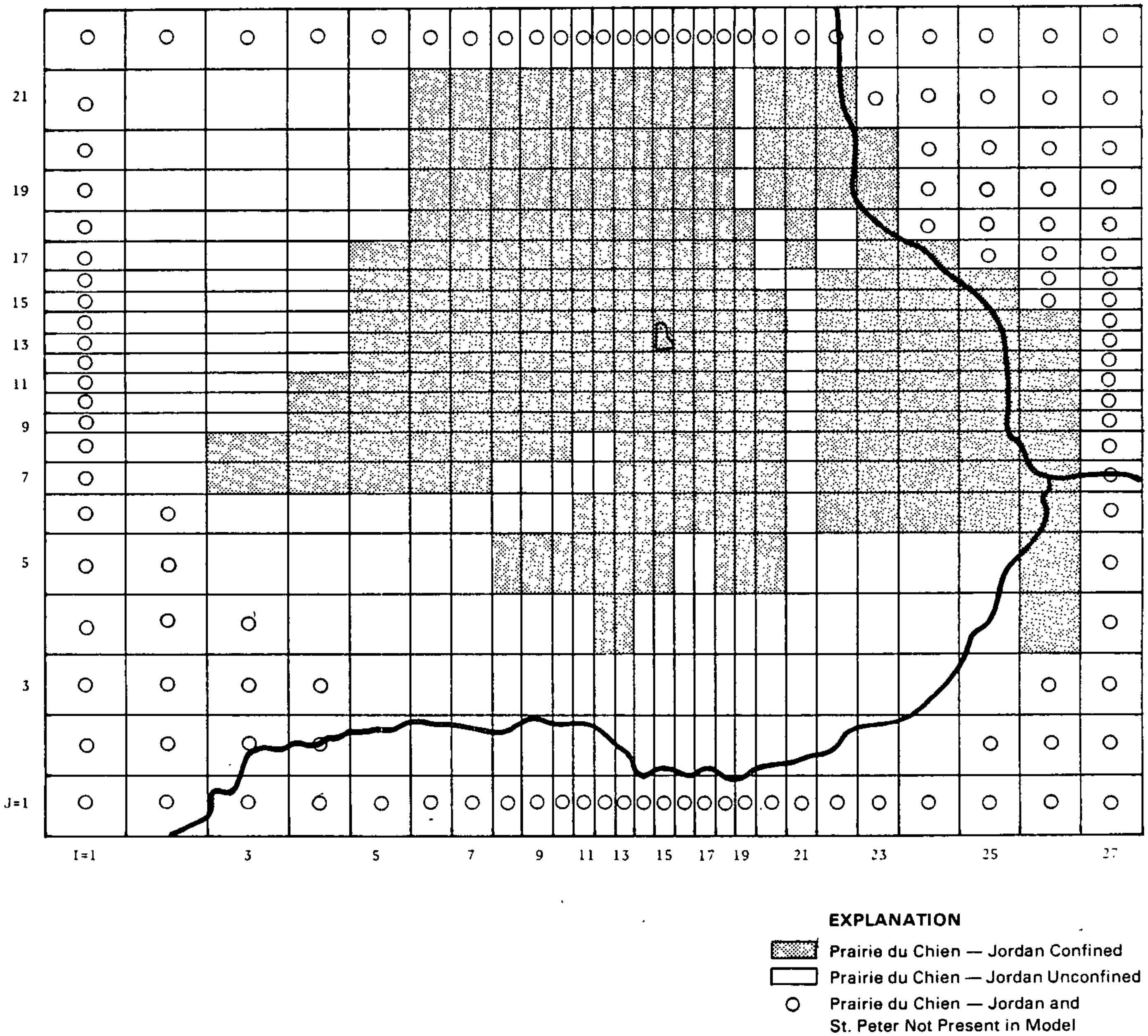

Figure E2-11 Model Finite-Difference Grid Showing Absence or Presence of Confining Bed Above Prairie du Chien-Jordan Aquifer

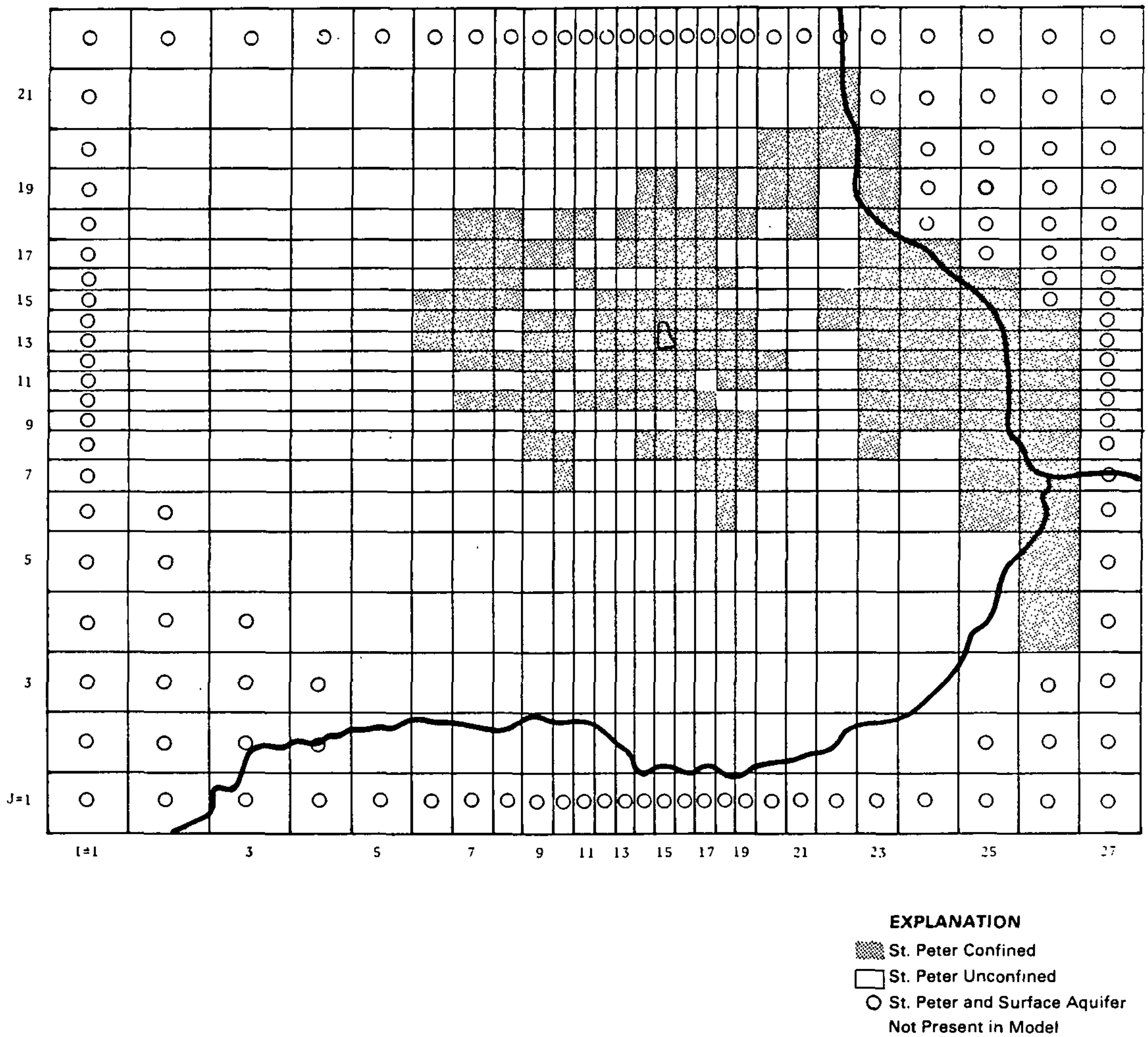

Figure E2-12 Model Finite Difference Grid Showing Absence or Presence of Confining Bed Above St. Peter Aquifer

TABLE E2-2

MAJOR GROUND-WATER PUMPAGE IN STUDY AREA

	Grid Location I	Grid Location J	Pumpage(1) gallons per minute	Pumpage(1) cubic feet per second
Mt. Simon-Hinckley Aquifer				
St. Louis Park 11	15	14	609	1.36
St. Louis Park 12	17	10	209	.47
St. Louis Park 13	16	16	196	.44
Edina 9	15	7	143	.32
Edina 10	19	5	495	1.10
Edina 12	13	10	111	.25
*	3	11	313	.70
*	2	11	269	.60
*	2	13	269	.60
*	2	15	224	.50
*	22	18	313	.70
*	23	18	269	.60
*	3	3	269	.60
Ironton-Galesville Aquifer				
none				
Prairie du Chien-Jordan Aquifer				
St. Louis Park 4	18	11	177	.39
St. Louis Park 5	14	13	292	.65
St. Louis Park 6	17	10	674	1.50
St. Louis Park 7	15	15	42	.10
St. Louis Park 8	12	16	724	1.61
St. Louis Park 9	15	15	42	.09
St. Louis Park 10	15	14	166	.37
St. Louis Park 14	16	16	197	.44
St. Louis Park 15	15	14	449	1.00
St. Louis Park 16	12	16	853	1.90
Edina 2	18	9	823	1.83
Edina 3	19	8	75	.17
Edina 4	17	8	323	.72
Edina 5	19	6	168	.37
Edina 6	17	7	672	1.50
Edina 7	16	7	140	.31
Edina 8	16	6	124	.28
Edina 11	19	5	766	1.71
Edina 13	13	10	744	1.66
Edina 14	14	5	89	.20
Edina 15	14	9	327	.73
Edina 16	14	6	521	1.32
Edina 17	18	6	226	.50

*Location and pumpage determined from Guswa et al. (1982)

TABLE E2-2 (Continued)

MAJOR GROUND-WATER PUMPAGE IN STUDY AREA

	Grid Location I	Grid Location J	Pumpage[1] gallons per minute	Pumpage[1] cubic feet per second
Prairie du Chien-Jordan Aquifer				
Edina 18	20	6	96	.21
Hopkins 1	11	10	519	1.16
Hopkins 3	13	11	137	.31
Hopkins 4	10	12	1107	1.24
Hopkins 5	11	12	1107	1.24
Minnetonka 9	7	15	201	.45
Plymouth 1,2,3	7	19	1120	2.50
Eden Prairie 1,2	8	5	449	1.00
Richfield 6	23	6	349	.78
Robbinsdale 5	19	20	295	.66
W29 (Flame Industry)	15	12	23	.05
W40 (Minn. Rubber)	16	12	158	.35
W45 & W46 (S-K Products)	17	12	5	.01
W48 (Methodist Hospital)	16	11	440	.98
W62 (McCourtney Plastics)	15	15	48	.13
W63 (National Foods)	16	15	128	.28
W80 (Red Owl)	12	12	157	.35
Dayton's 3	22	17	224	.50
Dayton's 4	20	6	896	2.0
NW National Bank 2	23	17	313	.70
General Mills 1	13	19	291	.65
NW Orient 1,2 and MSP Intl. Airport	25	6	1210	2.70
St. Peter Aquifer				
St. Louis Park 3	15	14	121	.27
W45 & W46 (S-K Products)	17	12	5	.01
W62 (McCourtney Plastics)	15	15	58	.13
Drift-Platteville Aquifer				
Wyzata 2	5	17	224	.50
Wyzata 3,4	5	18	336	.75
St. Louis Park 3	15	14	121	.27

(1) Pumpage rates based upon Hult and Schoenberg (1981) and Minnesota Department of Natural Resources (1982) except where noted.

$$V_x(i,j,k) = \frac{K_k}{n_k} \frac{(h_{i,j-1,k} - h_{i,j+1,k})}{0.5\ (\Delta x_{j-1} + 2\Delta x_j + \Delta x_{j+1})}$$

where,

- i is the x-direction node index,
- j is the y-direction node index,
- k is the layer index,
- V is ground-water flow velocity at the center of node (i,j,k) in the three-dimensional finite difference grid (L/T),
- K_k is hydraulic conductivity of layer k (L/T),
- n_k is porosity of layer k,
- h is hydraulic head at nodes on either side of (i,j,k) (L), and
- Δx is x-dimension length of nodes j, $j-1$, $j+1$ (L).

This equation is solved for each node in both the x and y directions in each layer. The computed velocities are printed with output from the USGS model.

Porosity, n, and hydraulic conductivity, K, must be specified as input to the model for each aquifer layer. The following values, obtained primarily from Norvitch et al. (1973), were used in the model:

	Porosity (-)	Hydraulic Conductivity (feet per second)
Mt. Simon-Hinckley	0.22	9.3×10^{-5}
Ironton-Galesville	0.25	1.2×10^{-5}
Prairie du Chien-Jordan	0.15	4.3×10^{-4}
St. Peter	0.28	1.4×10^{-4}
Platteville-Drift	0.30	7.5×10^{-4}

The velocity of ground-water flow is proportional to the ratio of K/n for a given hydraulic gradient. The use of average n and K values for a model layer is an approximation in layers composed of multiple formations between which K/n may vary. In the model of the St. Louis Park area there is only minor error associated with this approximation. For example, the value K/n for the Prairie du Chien Group is twice the value assumed for model layer 3, while the value for the Jordan Formation is one-half. These discrepancies are comparable to the range of uncertainty and variability in the field data used to specify n and K.

A velocity and travel path plotting program was written to present results from the USGS flow model in a more useful format. Velocity information computed by the modified USGS flow model is transferred to the plotting program via data file. The plotting program plots velocity diagrams indicating the direction and speed of the flow field. The program also plots travel paths beginning at selected points in the model. The travel path plotting option uses the ground-water flow velocity and direction at a node and the grid spacing to compute travel time and flow path across the finite difference grid. The node from which the travel path is to begin is specified by the user.

The velocity and travel path information is useful in the approximate determination of the direction and rate of movement of contaminants in the ground-water flow system, neglecting the influence of dispersion and physical/chemical processes.

E2.4 Model Calibration

The task of model calibration involves matching the computed head distribution of each layer with heads measured at various points in the field and with published potentiometric surface maps. This is performed by adjustment of input data within reasonable ranges based on field data.

Several simulations of the flow model were made before an acceptable calibration was achieved for each layer. Figures E2-13 through E2-16 show a comparison of potentiometric surface contours drawn from model-computed heads with potentiometric surface maps taken from existing literature for each aquifer layer except the Ironton-Galesville. No potentiometric surface maps were available to this study for the Ironton-Galesville due to the scarcity of wells penetrating this aquifer. The modeled and observed extent of the St. Peter is also shown on Figure E2-15.

In general, the calibration is good for the Mt. Simon-Hinckley, Prairie du Chien-Jordan, and Drift-Platteville aquifers. Minor problems with computed heads in the northwest corner of the grid may be due to localized variations in aquifer charcteristics, large nodal spacing, or inadequate representation of flow from outside the model boundary.

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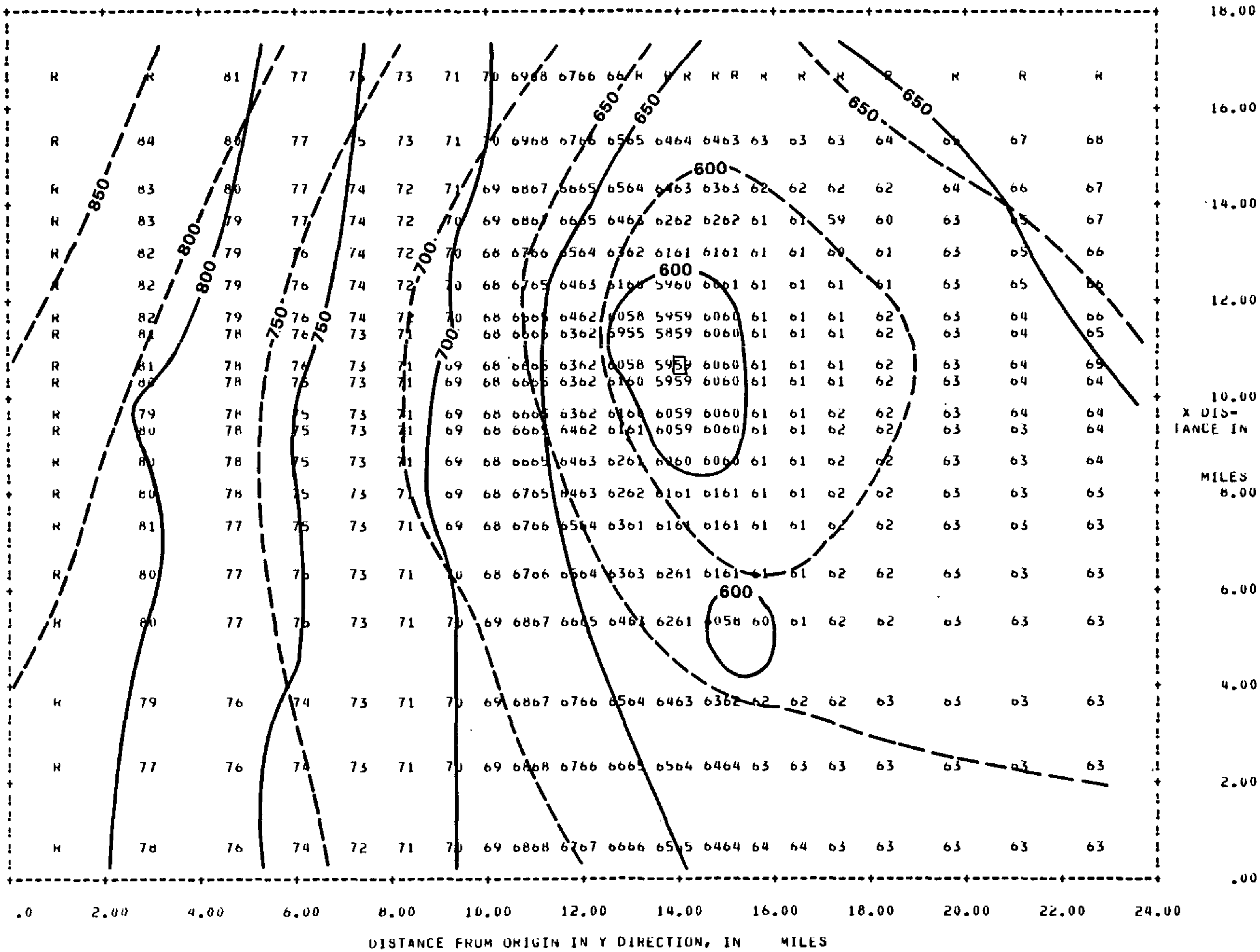

EXPLANATION

— — — Observed (Norvitch et al. 1973)

——— Model-Computed

Numbers show computed hydraulic head in 10's of feet, n.g.v.d.

Figure E2-13 Comparison of Calibrated Model Results with Published Potentiometric Surface Data for Model Layer 1 (Mt. Simon-Hinckley Aquifer)

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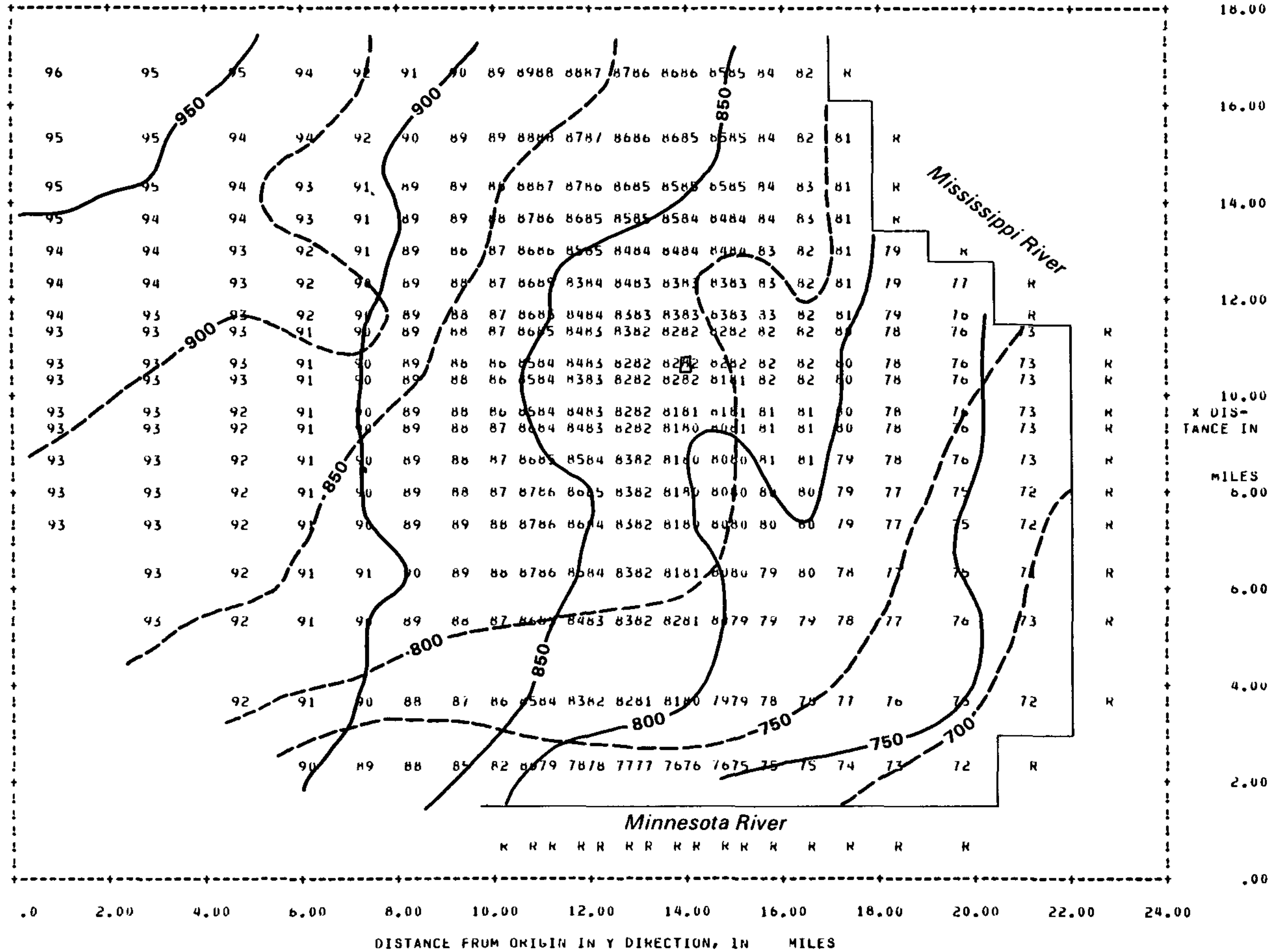

EXPLANATION

Observed (Hult and Schoenberg 1981 and Norvitch et al. 1973)

Model-Computed

River or Lake Boundary

Numbers show computed hydraulic head in 10's of feet, n.g.v.d.

Figure E2-14 Comparison of Calibrated Model Results with Published Potentiometric Surface Data for Model Layer 3 (Prairie du Chien-Jordan Aquifer)

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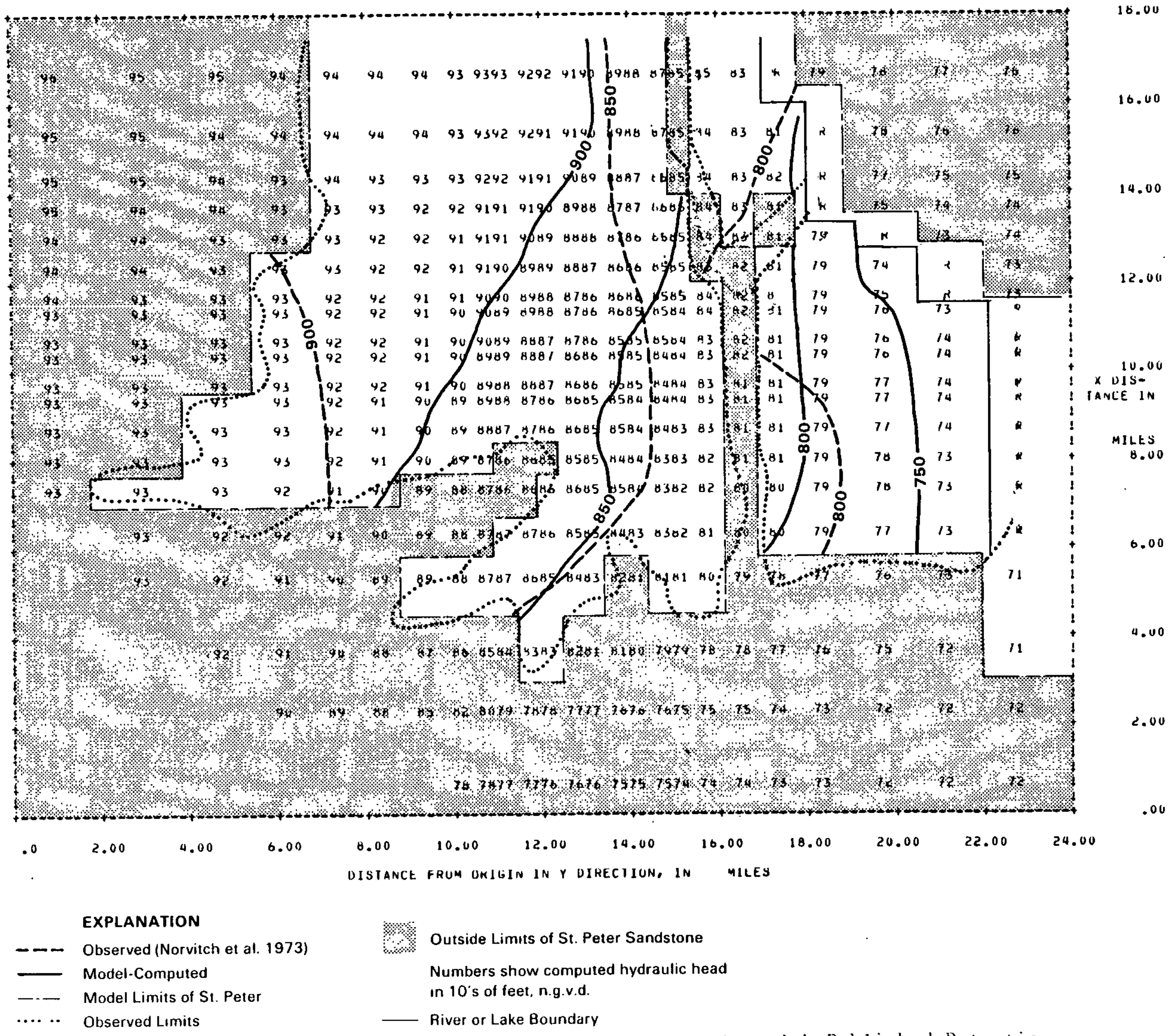
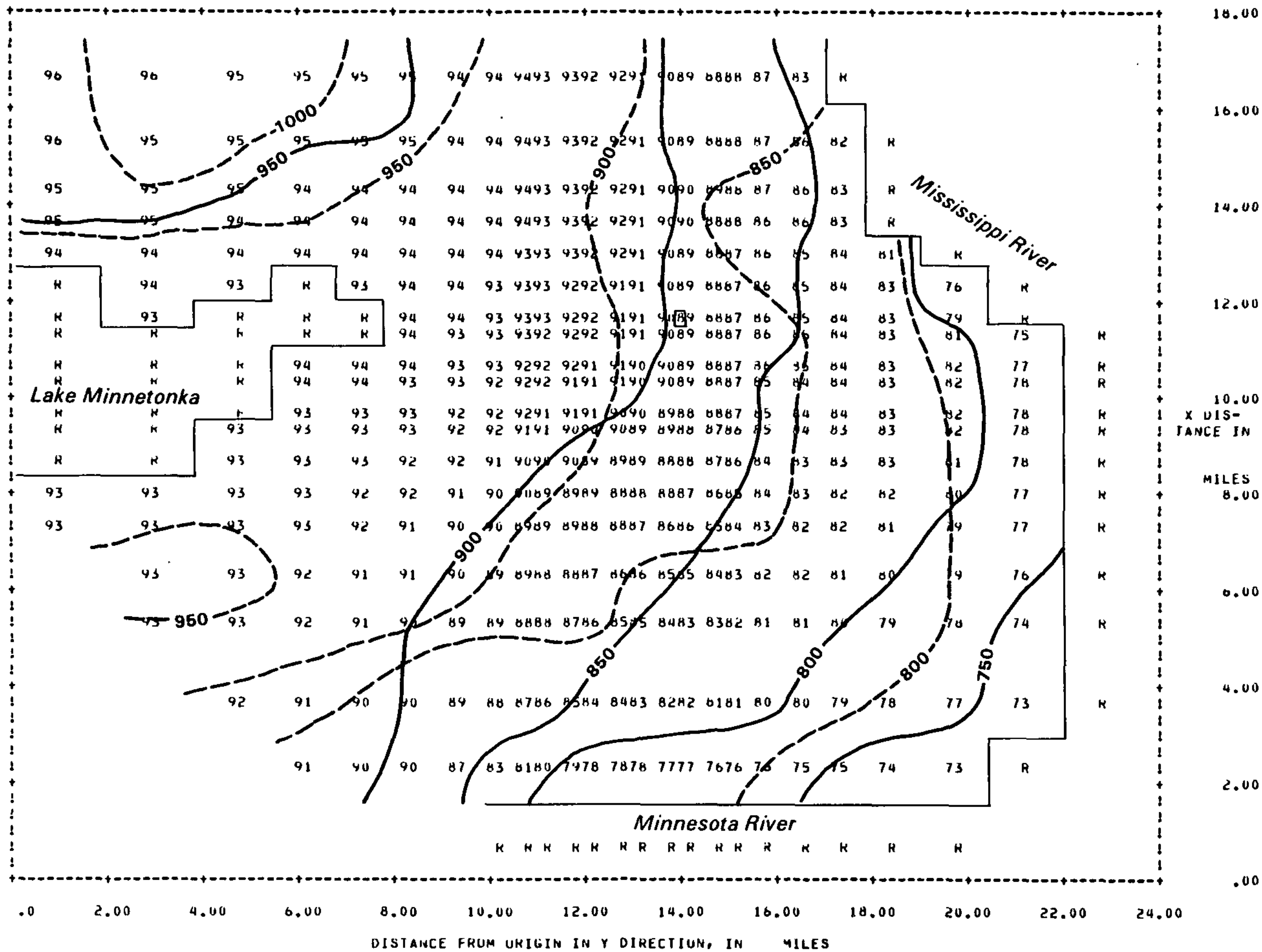


Figure E2-15 Comparison of Calibrated Model Results with Published Potentiometric Surface Data for Model Layer 4 (St. Peter Aquifer)



EXPLANATION

- Observed (Larson-Higdem et al. 1975)
- Model-Computed
- River or Lake Boundary

Numbers show computed hydraulic head in 10's of feet, n.g.v.d.

Figure E2-16 Comparison of Calibrated Model Results with Published Water Table Elevation Data for Model Layer 5 (Drift-Platteville Aquifer)

Calibration is reasonably good for the St. Peter aquifer (layer 4). The inherent problems associated with modeling this aquifer as both confined and unconfined and the presence of the bedrock valley were considered in the determination of an acceptable calibration for the St. Peter.

Calibration of the flow model compares favorably with that obtained by Guswa et al. (1982). In that analysis, performance of the three-dimensional model for an area somewhat larger than this study area is evaluated by comparing the difference between model-computed and observed potentiometric levels, which are termed "residuals". These residuals, which may be as great as 100 feet, are given for all aquifers except the Ironton-Galesville where simulated heads and 1971-1977 potentiometric levels are compared. Although the calibration of Guswa et al. (1982) is only preliminary, he primarily attributes poor model performance to inadequate representation of ground-water withdrawals.

Overall, the model calibration is quite satisfactory for the purposes of this study. The calibrations preserve potentiometric surface gradients and thus the general character of flow observed in the field. Particularly near the site area, ground-water flow directions and speeds will be in the adequate agreement between the model results and the observed behavior.

E2.5 Sensitivity Analysis

The sensitivity of computed head distributions to adjustment of certain parameter values was determined as part of the modeling study. Those parameters which were evaluated were transmissivity (for the Mt. Simon-Hinckley, Prairie du Chien-Jordan, and St. Peter aquifers), interaquifer leakage coefficient (between all aquifers), hydraulic conductivity (for the unconfined Drift-Platteville aquifer), and areal recharge from precipitation. Values for transmissivity, hydraulic conductivity, and areal recharge were adjusted ten per cent. Leakage coefficients were adjusted by a factor of ten.

Table E2-3 summarizes the change in computed head values at the node immediately north of the caused by the aforementioned changes in

TABLE E2-3

RESULTS OF SENSITIVITY ANALYSIS

Simulation Conditions	Layer	Head at Node (15,14)[1] (feet, n.g.v.d.)	Change in Head[2] (feet)
Model as calibrated	L1	551.28	
	L2	821.45	
	L3	821.49	
	L4	857.38	
	L5	906.22	
Increase Mt. Simon-Hinckley (L1) transmissivity 10%	L1	566.07	+14.79
	L2	821.67	+ .22
	L3	821.53	+ .04
	L4	857.41	+ .03
	L5	906.24	+ .02
Increase Prairie du Chien-Jordan (L3) transmissivity 10%	L1	551.09	- .19
	L2	821.62	+ .17
	L3	821.97	+ .48
	L4	856.09	- 1.29
	L5	904.93	- 1.29
Increase St. Peter (L4) transmissivity 10%	L1	551.23	- .05
	L2	820.97	- .48
	L3	821.00	- .49
	L4	857.37	- .01
	L5	905.54	- .68
Increase Platteville-Drift (L5) hydraulic conductivity 10%	L1	551.15	- .13
	L2	820.12	- 1.33
	L3	820.13	- 1.36
	L4	855.64	- 1.74
	L5	903.32	- 2.90
Increase leakage coefficient between Mt. Simon-Hinckley (L1) and Ironton-Galesville (L2) ten times	L1	635.81	+84.53
	L2	800.52	-20.93
	L3	817.33	- 4.16
	L4	854.82	- 2.56
	L5	904.58	- 1.64
Increase leakage coefficient between Ironton-Galesville (L2) and Prairie du Chien-Jordan (L3) ten times	L1	551.50	+ .22
	L2	821.85	+ .40
	L3	821.49	0
	L4	857.39	+ .01
	L5	906.24	+ .02
Increase leakage coefficient between Prairie du Chien-Jordan (L3) and St. Peter (L4) ten times	L1	551.72	+ .44
	L2	833.86	+12.41
	L3	834.17	+12.68
	L4	840.53	-16.85
	L5	899.92	- 6.30

TABLE E2-3 (Continued)

RESULTS OF SENSITIVITY ANALYSIS

Simulation Conditions	Layer	Head at Node (15,14)(1) (feet, n.g.v.d.)	Change in Head(2) (feet)
Increase leakage coefficient between St. Peter (L4) and Drift-Platteville (L5) ten times	L1	552.96	+ 1.68
	L2	829.95	+ 8.50
	L3	830.10	+ 8.61
	L4	870.03	+12.65
	L5	894.94	-11.28
Decrease areal recharge over Drift-Platteville (L5) 10%	L1	550.62	- .66
	L2	815.44	- 6.01
	L3	815.39	- 6.10
	L4	850.43	- 6.95
	L5	897.76	- 8.46

(1) Predicted potentiometric surface elevation at model node (15,14) with the specified simulation conditions.

(2) Difference between the potentiometric surface elevation at node (15,14) in the calibration run and the elevation determined with the specified simulation conditions. The elevations in the calibration run are given as the first entries in the table.

parameter values. This node was selected for presentation of the sensitivity study results because it is located in the model area of greatest interest to this study. To give a perspective on the head changes shown in Table E2-3, the seasonal head variations observed by Hult and Schoenberg (1981) near the site are 2 to 3 feet in the middle drift aquifer, approximately 7 feet in the Prairie du Chien-Jordan and 12 feet in the Mt. Simon-Hinckley.

With regard to 10 per cent increases in transmissivity and hydraulic conductivity in layers 1, 3, 4, and 5, head values in the Mt. Simon-Hinckley (layer 1) are most sensitive to parameter changes in that layer. In each case, however, head values in a layer are only slightly affected by parameter changes in another layer.

The model is relatively sensitive to order of magnitude increases in leakage coefficients, except for the Ironton-Galesville (layer 2) to Prairie du Chien-Jordan (layer 3) leakage. The greatest sensitivity to a ten-fold change in leakage coefficient is observed when the adjustment is made between the Mt. Simon-Hinckley and Ironton-Galesville aquifers. It is important to note that the predictive capabilities of the model are least utilized for these two layers.

Decreasing the areal recharge 10 per cent decreases head values in layers 2 through 5. Thus, the model is relatively sensitive to changes in recharge.

Although changes in the value of head at an indicator node, as presented in Table E2-3, are useful in showing the relative model sensitivity, it is also important to relate the sensitivity study results to the purposes of the model. In this case, hydraulic transport is of primary interest, particularly in the Prairie du Chien-Jordan aquifer. Figure E2-17 illustrates the sensitivity of the model flow predictions to selected parameter changes. The indicator variable used in Figure E2-17 is the eastward velocity component along a west to east transect that includes the plant site model node. The predicted velocity is relatively insensitive to the areal recharge and moderately sensitive to the layer transmissivity. Greater sensitivity is found with respect to

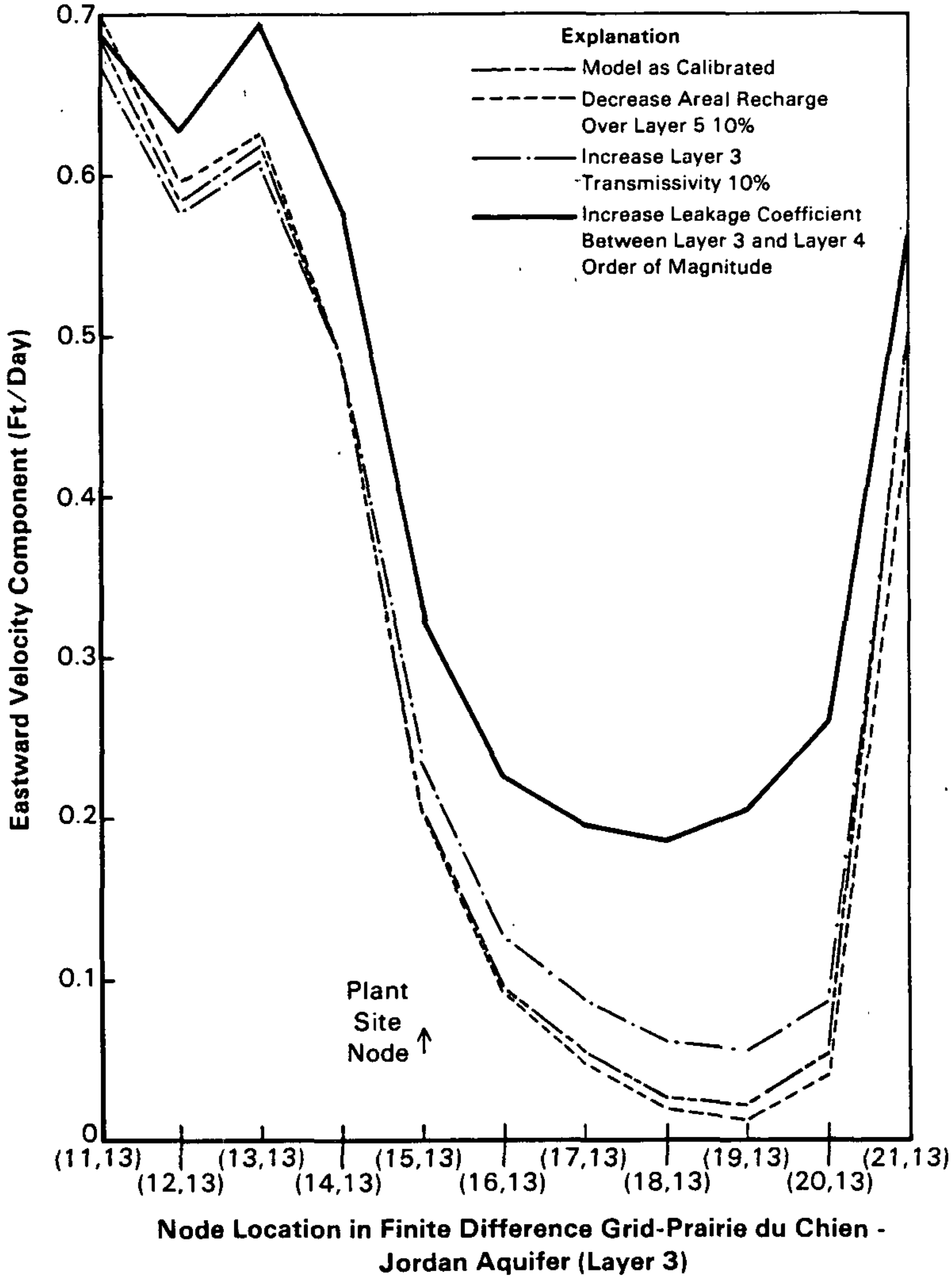


Figure E2-17 Computed Velocities for Sensitivity Analysis of Selected Parameters

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the inter-layer leakage coefficient. However, this coefficient also proved fairly sensitive in the evaluations based on head in Table E2-3, thus greatly erroneous leakage parameters would significantly impact the model calibration which is based on head. Although significant sensitivity exists, major errors in the selected parameters values is unlikely given the generally good model calibration results. Overall, the sensitivity of the model predictions is relatively small given that parameters may be varied only within certain bounds to still achieve acceptable calibration. Few of the sensitivity runs shown in Table E2-3 would achieve acceptable calibrations and none are as good as the parameter set associated with the model as calibrated.

Model sensitivity is ultimately important to the extent that it impacts the results of the model upon which study conclusions are based. The sensitivity studies show that drastic sensitivity -- for example, a reverse in flow direction -- is nowhere observed. Rather, the predicted flow velocities appear to be relatively insensitive to parameter variations. Thus the conclusions of this study as to contaminant migration direction and speed would not change appreciably due to reasonable variation of the model parameters. Sensitivity to greater change in the parameters is not a concern since such changes would destroy the model calibration. The joint conclusion of the calibration and sensitivity efforts is that the model may be used with confidence as a predictive tool for study of the St. Louis Park problem.

E2.6 Confined Aquifer Analytical Model

The USGS three-dimensional flow model, as programmed for this study, does not resolve features of ground-water flow on a scale of less than one-half mile. This scale is quite adequate for the analyses required in this study, with the exception of small scale effects associated with the many closely-spaced wells in the Prairie du Chien-Jordan aquifer. To analyze small scale features, an additional model was developed.

The detailed scale model is based upon the analytical solution of the differential equation for a leaky confined aquifer. The solution is that given by Bear (1979) for the steady-state drawdown about a well in an isotropic confined aquifer:

$$s = \phi_o - \phi \approx \frac{Q_w}{2\pi T} \quad K_o(r/\lambda)$$

where,

s	is the drawdown at radius r (L),
ϕ	is the potentiometric surface elevation in the confined aquifer (L),
ϕ_o	is the potentiometric surface elevation in the adjacent aquifer connected by leakage (L),
Q_w	is the rate of pumping at the well (L^3/T),
T	is the aquifer transmissivity (L^2/T),
r	is the radial distance from the well (L),
λ	is the confining bed leakage factor (L) as defined by Bear (1979), and
K_o	is the Modified Bessel function of the second kind of order zero.

The model computes the potentiometric surface at a point by superposing the drawdown due to all wells included in the simulation. Both pumping and recharge wells may be included and a linear gradient in ϕ_o may be considered as well. The analytical model was programmed in BASIC computer language and operated on a micro-computer.

In the application to the Prairie du Chien-Jordon aquifer, the analytical confined aquifer model was utilized with a value of transmissivity of 56,000 gallons per day per foot and coefficient for leakage through the basal St. Peter confining bed of 1.25×10^{-10} per second. Leakage from the Prairie du Chien-Jordan to the underlying layer was neglected; however, that leakage is an order of magnitude less than the St. Peter to Prairie du Chien-Jordan leakage. Computed heads were found to be in reasonable agreement with those observed in the field.

E2.7 Contaminant Retardation Effects

It is well known that the velocity at which ground water moves may be much greater than the velocity at which dissolved contaminants travel (Freeze and Cherry 1979). Dissolved contaminants are carried with the water and thus travel in the same direction. However, contaminants may undergo chemical reactions or physical transformations and be diluted by mixing (dispersion). In this appendix, travel paths and travel times are predicted according to the predicted hydraulic transport, the transport experienced by a particular minuscule packet of water. This transport can be predicted far more reliably than contaminant transport, thus its use in this study. However, this is a conservative approach since contaminants will generally travel tenfold to one-hundredfold more slowly. Thus, the predictions in this appendix are pessimistic in the sense that the velocity at which contaminants spread is likely to be exaggerated.

Information is limited; however, there is some basis to estimate the velocity of contaminant travel relative to that of hydraulic transport. A common model for contaminant transport in ground water assumes that there is a simple multiplicative relation between the ground-water velocity, V, and the contaminant velocity, V_c (Freeze and Cherry 1979). The multiplicative constant is known as the retardation factor, R_d, defined as:

$$R_d = \frac{V}{V_c} = 1 + \frac{\rho_b}{n} \cdot K_d$$

where,

ρ_b is the bulk mass density of the porous medium (M/L^3),

n is the medium porosity, and

K_d is the distribution coefficient (L^3/M), which is equal to the mass of contaminant on the solid phase per unit mass of solid phase divided by the concentration of contaminant in solution.

If the necessary parameters can be determined or estimated, an approximate value of the retardation factor can be computed and used to adjust values of contaminant velocity (or inversely, travel time). Estimates of retardation factors for various aquifers in St. Louis Park are given in the following paragraphs.

Table E2-4 lists values taken from available literature which were used to determine a value for the retardation factor for PAH in selected geologic units in the St. Louis Park area. An approximate retardation factor of 20 to 25 is computed for the Drift-Platteville and St. Peter aquifers.

Values for the distribution coefficient shown in Table E2-4 are taken from the results of research performed at the Department of Geology and Geophysics at the University of Minnesota (Cohen 1982). Cohen determined the distribution coefficient in laboratory experiments performed with field samples of the aquifer materials. Values for the distribution coefficient shown in Table E2-4 are from batch tests using naphthalene. These values are considered to be more indicative of site conditions than those given by Hickok and Associates (1981) based on literature values. Hickok gives distribution coefficient values of 60 to 168 cubic centimeters per gram for "five carcinogenic PAH" and of 5 to 23 cubic centimeters per gram for "five other PAH". Using a range of 60 to 168 cubic centimeters per gram for the distribution coefficient for the Drift unit, the retardation coefficient would be 19 to 52 times greater than the value given in Table E2-4. Hence, the values used in this study are assumed to be realistic and valid, but conservative.

No retardation factor is given in Table E2-4 for the Prairie du Chien Group because its solution channel geology creates entirely different contaminant retardation effects than a granular porous medium. There is insufficient data available for the Prairie du Chien to define the retardation factor. Further, the aquifer's solution channel geology makes estimation based on literature values difficult. For example, literature values for the retardation factor of radioactive substances in dolomite vary by as much as a factor of

TABLE E2-4

VALUES USED IN DETERMINATION OF RETARDATION FACTORS

Geologic Unit	Bulk Mass Density[a] (grams per cubic centimeter)	Porosity[c]	Distribution Coefficient[d] (cubic centimeters per gram)	Retardation Factor
Drift	1.86[b]	0.30	3.13	20
St. Peter	2.02	0.28	2.74	21

[a]Norvitch and Walton 1979.
[b]For mixed-grained sand, dense. Terzaghi and Peck 1967.
[c]Norvitch et al. 1973.
[d]Cohen 1982.

25 for particular elements and by many times more between different elements (Mercer et al. 1982). In the absence of usable field data or literature values, the Prairie du Chien-Jordan will be assumed to have zero PAH retardation in this study.

The retardation factors defined in the preceding discussion are based upon laboratory studies of a single PAH compound, naphthalene. Unfortunately, data of similar specificity do not exist for other PAH compounds; indeed, there are few data on retardation in ground water for PAH compounds of any sort. Nevertheless, it is possible to draw qualitative conclusions on the retardation of different PAH compounds from published studies of PAH sorption in river and lake sediments and in soil (Karickhoff et al. 1979 and Means et al. 1980). The conclusion from those studies are that higher molecular weight PAH compounds, and particularly those PAH compounds with the greatest molecular chain length, are more highly sorbed to the solid phase than lower molecular weight PAH. Higher adsorption leads to a greater retardation factor. This implies that the retardation factors presented in Table E2-4 are conservatively low, since they apply to the low molecular weight, two-ring molecule, naphthalene. Further, the conclusions imply that carcinogenic PAH, which are heavy and long molecules, will be more greatly retarded than most non-carcinogenic PAH.

The experiments by Karickhoff et al. and Means et al. quantify adsorption in terms of K_{oc}, the partition coefficient expressed per fractional organic content:

$$K_{oc} = \frac{K_d}{oc}$$

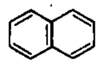
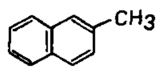
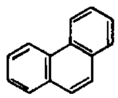
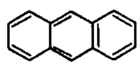
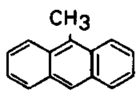
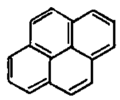
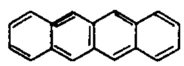
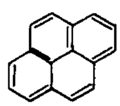
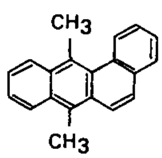
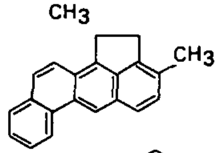
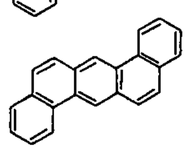
where oc is the fractional mass of organic carbon in the sediment or soil (-), and

K_{oc} is the distribution coefficient normalized by organic content (L^3/M).

The variation of K_{oc} is a good indicator of how K_d is likely to vary in porous media. Table E2-5 shows the variation of K_{oc} for different PAH compounds. It is clear that adsorption varies strongly

TABLE E2-5

VARIATION OF PAH ADSORPTION VERSUS MOLECULAR STRUCTURE

Compound	Molecular Structure	Molecular Weight	Adsorption Coefficient K_{oc} (mℓ/g)
Adsorption on Pond Silt (Karickhoff et al. 1979)			
Naphthalene		128	1.3
2-methylnaphthalene		142	8.5
Phenanthrene		178	23
Anthracene		178	26
9-methylanthracene		192	65
Pyrene		202	84
Tetracene		228	650
Adsorption on Sediments and Soils (Means et al. 1980)			
Pyrene		202	63
7,12-dimethylbenz(a)anthracene		256	236
3-methylcholanthrene		268	1789
Dibenz(a,h)anthracene		278	2029

with molecular size; for example, four-ring tetracene is adsorbed 500 times more strongly than double-ringed naphthalene. Retardation coefficients can be expected to be related in approximately the same proportion. Thus, the direct conclusion of Table E2-5 is that the redardation factor used in this study is conservatively small when applied to larger-ringed PAH compounds. The degree of this conservatism is in excess of 1000-fold for the largest (five-ring and six-ring) compounds.

E3. SIMULATION RESULTS

E3.1 Introduction

This section reports on the results of simulations designed to evaluate the behavior of the ground-water system in the area of St. Louis Park. The purpose of the simulations discussed in this section is to analyze and understand the prevailing ground-water transport pattern. The information gained in this section is utilized to propose appropriate future actions in response to the contamination of ground water in St. Louis Park.

The sub-sections which follow present the findings for individual aquifers in the St. Louis Park subsurface. The order of presentation is by depth, beginning with the uppermost aquifer.

E3.2 Analysis of the Drift-Platteville Aquifer

For the purposes of flow modeling, the upper, middle and lower drift and the Platteville Limestone were taken as a single hydrogeologic unit. Water-table conditions prevail in this unit, which is influenced by infiltration recharge, leakage to underlying layers, and pumping. Pumping is widely distributed in numerous small wells. There are few major wells screened in this unit, with the notable exception of St. Louis Park municipal supply well No. 3 (SLP3) which is open to both the Platteville limestone and St. Peter sandstone.

Highly detailed data on the water-table elevation in the surficial aquifer are available from Larson-Higdem et al. (1975). The data show gradients of approximately 16 feet per mile tending in a southeasterly direction towards the Mississippi and Minnesota Rivers. This regional trend is well duplicated in the results of computer simulations performed in this study using the USGS flow model. (See Section E2.5)

The computer model results were used to assess the direction and rate of transport in the Drift-Platteville aquifer (Figure E3-1). With the exception of the area influenced by Lake Minnetonka, transport is predicted to be directed to the east and south at

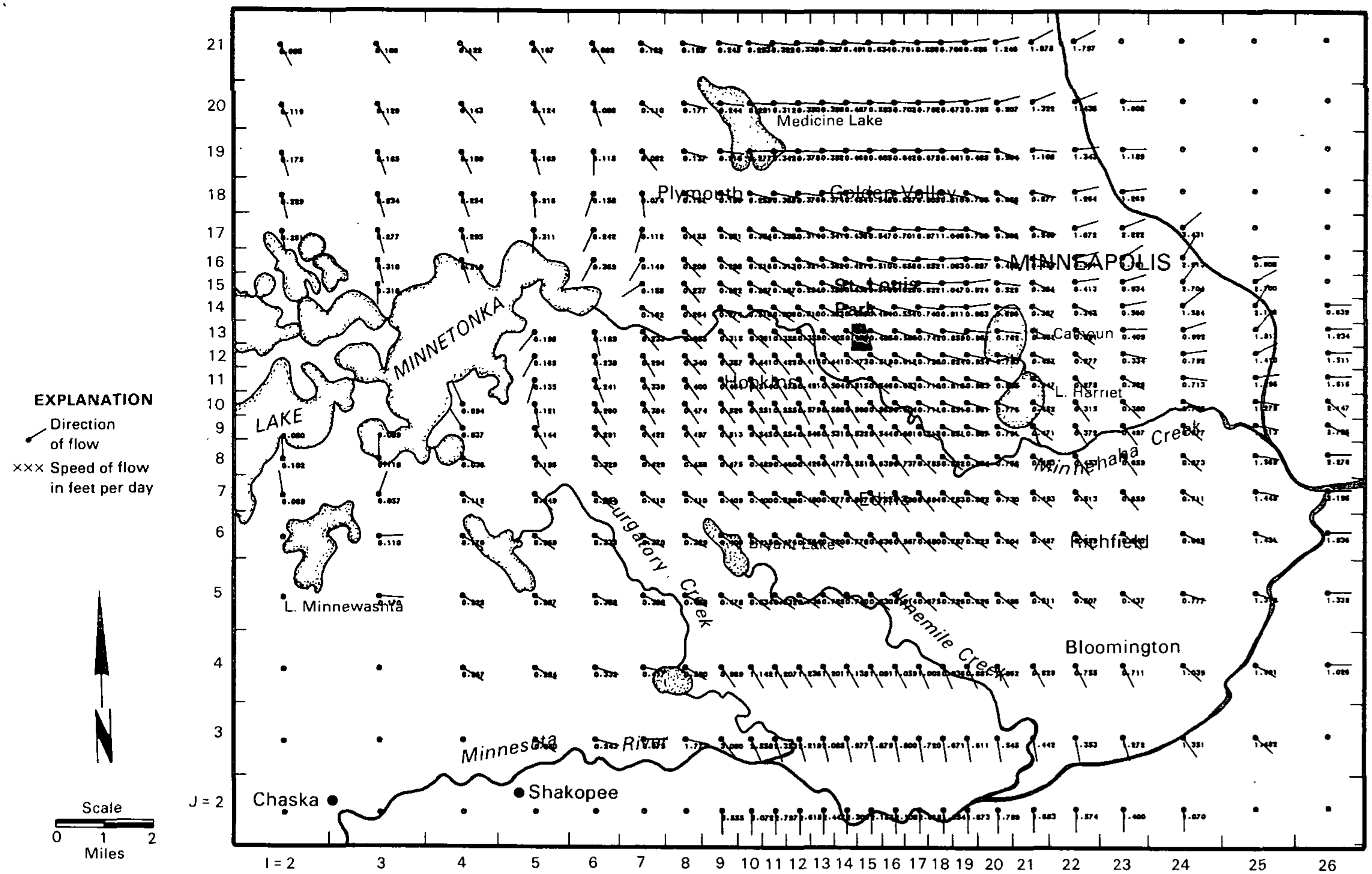


Figure E3-1 Pattern of Ground-Water Flow in the Drift-Platteville Aquifer Predicted by Computer Model velocities of roughly 0.5 feet per day. This speed is a conservative approximation of the speed at which contaminant would also be transported, neglecting the slowing influences of dispersion and physical/chemical retardation.

Figure E3-2 depicts the computer model predictions of the path and time of travel of ground water originating at selected points at and adjacent to the plant site. The paths follow the predicted ground-water flow patterns to their ultimate destinations at the river boundaries. The paths and travel times are those that a parcel of water would follow, assuming the regional flow pattern remained constant through time. Contaminant would follow the same general path, however contaminant would be diluted by dispersive mixing and would be slowed in its travel by chemical and physical retardation. The travel paths thus illustrate the likely directions of contaminant migration and conservatively underestimate the time required for contaminant to reach various locations.

Also shown on Figure E3-2 is the approximate location of the buried bedrock valley associated with an ancient river channel underlying the Minneapolis Chain of Lakes. The buried valley cuts through the Glenwood shale confining bed to the St. Peter aquifer and, over a narrower path, through the basal St. Peter to the Prairie du Chien Group. The buried valley is filled with glacial drift and till, and thus represents an area in which the Drift-Platteville aquifer communicates hydraulically with the St. Peter and Prairie du Chien aquifers. An important detail of the subsurface structure is the small tributary bedrock valley south of the plant site. Adequate potentiometric data for the St. Peter and Drift-Platteville aquifers is not available to evaluate the importance of this feature to ground-water flow in the adjacent aquifer units. The potentiometric information for the surface formations (Hult and Schoenberg 1981) shows little influence due to the buried bedrock valley opening other than in the basal drift immediately adjacent to the bedrock valley. Significant induced horizontal transport is not apparent in the field data. The bedrock valley was approximated in the computer model as an area of increased leakage from the surface layer to the St. Peter aquifer.

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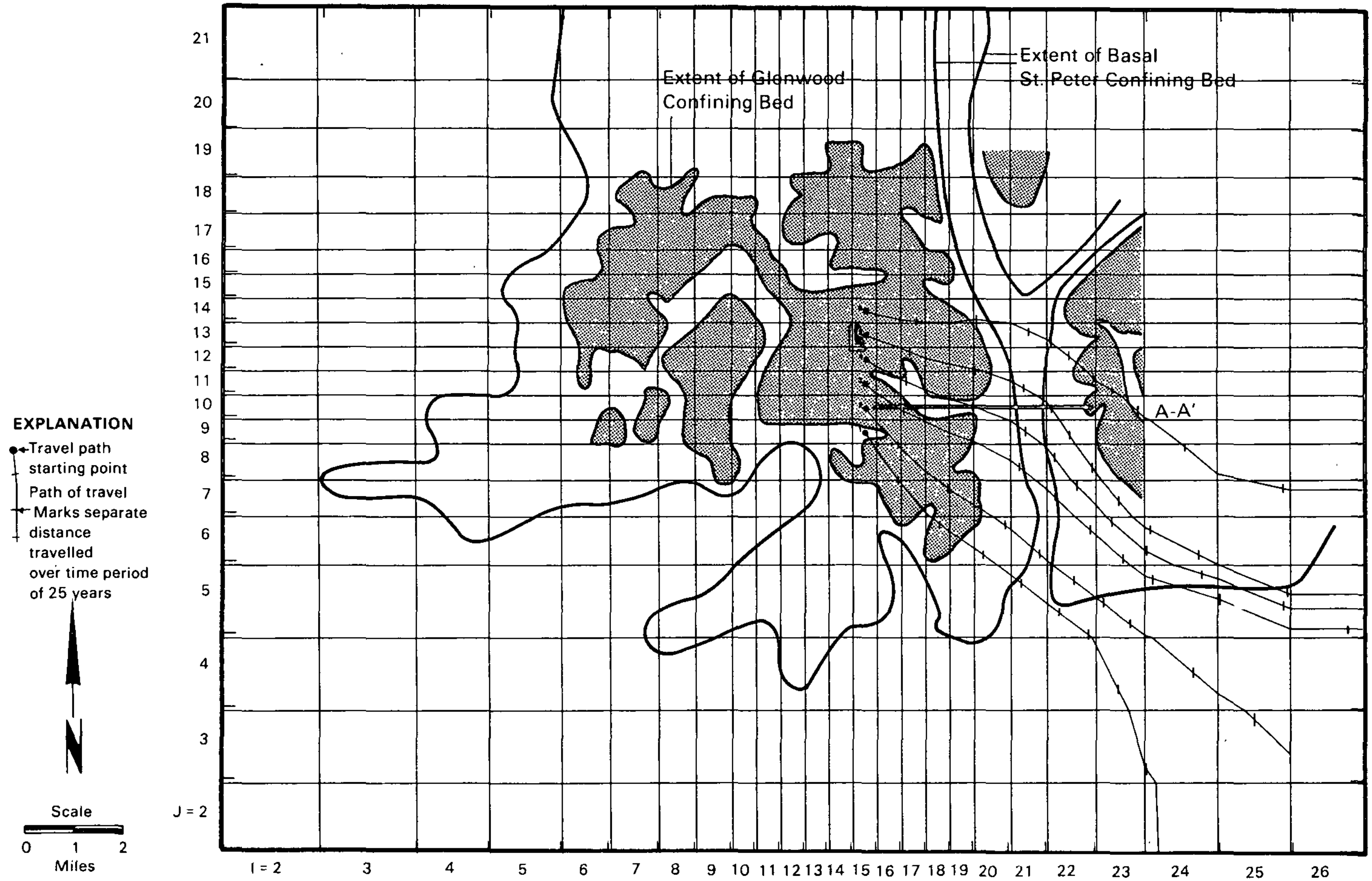

Figure E3-2 Selected Hydraulic Transport Travel Paths in the Drift-Platteville Aquifer Predicted by Computer Model

The path of travel shown in Figure E3-2 predicts transport from the site and its environs to the tributary bedrock valley within one or two decades. The travel path then proceeds on to the main bedrock valley and eventually to the Mississippi River. Travel times to the Prairie du Chien contact are on the order of sixty to seventy years, while transport to the Mississippi River requires centuries. These predictions cannot be taken to be exact, but merely as indicators of the scale of time required to reach these destinations. They are conservative in the sense that physical and chemical retardation will act to slow the advance of contaminant, lengthening the time of travel to greater than that predicted.

The influence of vertical transport through the bedrock valley is indicated on Figure E3-3. The figure shows the leakage along section AA' located in Figure E3-2. The section includes both areas within and without the buried bedrock valley. Leakage through the bedrock valley is computed based upon a leakage coefficient of 9.3×10^{-9} per second. This value assumes a layer of till roughly 2.5 feet thick with the vertical hydraulic conductivity of .002 feet per day indicated by Larson-Higdem et al. (1975). Vertical flow velocity components are predicted to be at least one order of magnitude less than those in the horizontal, even within the bedrock valley. Thus, although the bedrock valley can act as a flow conduit from the Drift-Platteville aquifer to the St. Peter, it appears unlikely from the field data available and from computer simulation results that the valley acts as an appreciable influence on the piezometric surface and horizontal flows in the neighboring Drift-Platteville aquifer.

The influence of the bedrock valley on flow in the Drift-Platteville is summarized as follows. Computer results show that a portion of the horizontal flow in the Drift-Platteville above the bedrock valley is intercepted and flows vertically down into the valley to the St. Peter aquifer. However, this is a relatively local influence: the valley does not create a notable depression in the surrounding water table nor does it induce significant horizontal flow about the valley. This is confirmed in the detailed field observations of the water table published by Larson-Higdem et al. (1975) which do not indicate a noticable depression in the locale of the bedrock valley.

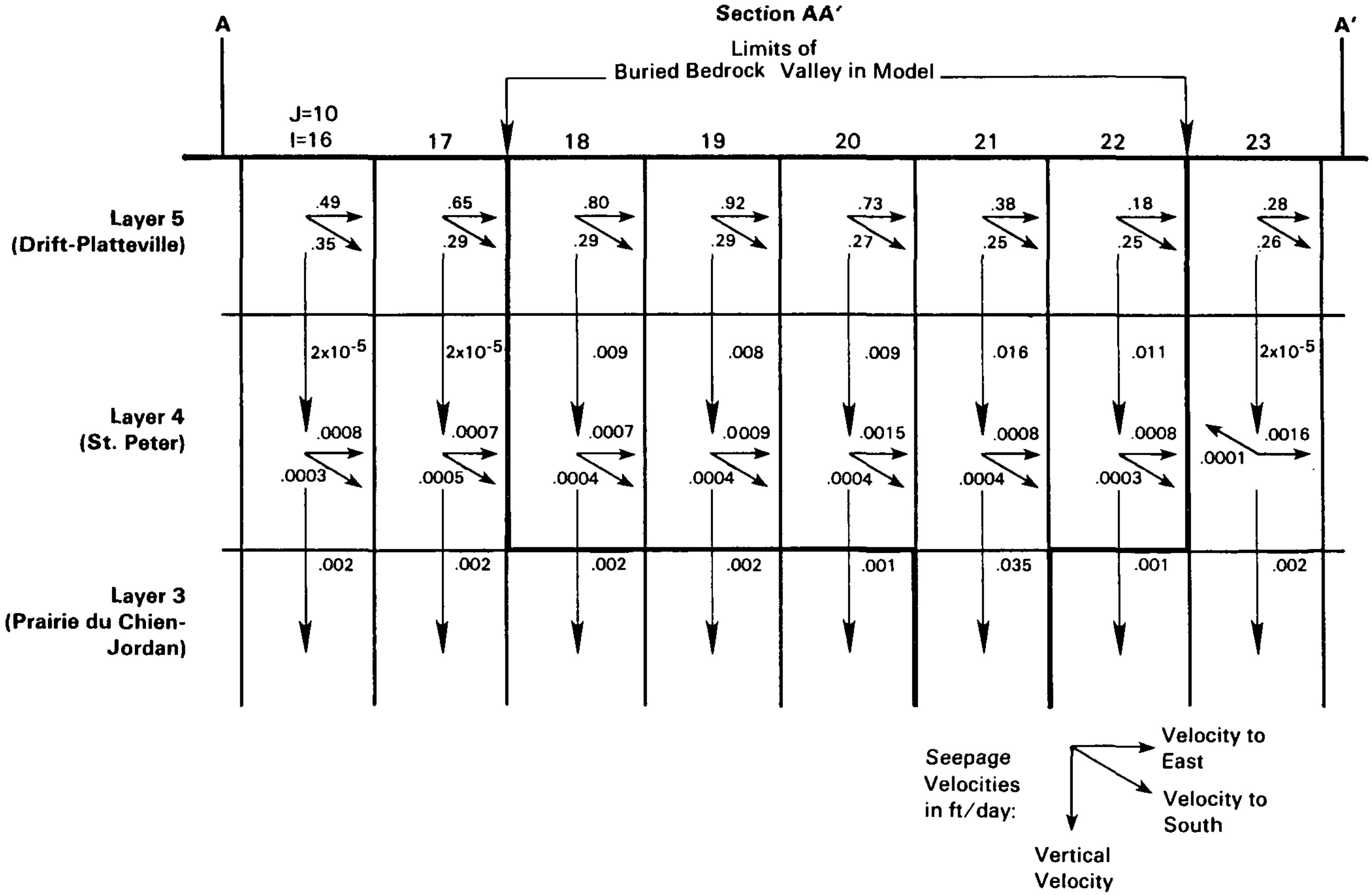

Figure E3-3 Horizontal and Vertical Flow Velocities at Buried Bedrock Valley as Determined in Computer Simulations

The single major well pumping from the Drift-Platteville in the vicinity of the site is SLP3. Computer simulations make the assumption that one-half of the flow to this well comes from each of the Platteville and St. Peter aquifers. In the simulations, well SLP3 was found to have no appreciable influence on flow in the Drift-Platteville when the well was operated at average pumping rates. When SLP3 was simulated to be operating at full capacity, the simulated water table was lowered about the well; however, the predicted drawdown was insufficient to cause flow of ground-water from the site area to SLP3. The time of travel for ground water to travel from the site is predicted to exceed 70 years, however. Accounting for retardation effects, the travel time exeeds 1400 years. Contamination of SLP3 from the Drift-Platteville is thus a remote possibility.

E3.3 Analysis of the St. Peter Aquifer

The St. Peter is a relatively minor aquifer for ground-water supply in the St. Louis Park area. The single major withdrawal well within the area is St. Louis Park municipal supply well No. 3 (SLP3). There are also a number of active industrial supply wells.

E3.3.1 Regional Flow

There is less information published describing the potentiometric surface of the St. Peter than is available for the Drift-Platteville or Prairie du Chien-Jordan aquifers. The data of Norvitch et al. (1973) indicate a regional pattern characterized by a gradient from west to east in a nearly due-east direction. The hydraulic gradient varies from about 5 to 15 feet per mile in the potentiometric surface maps presented by Norvitch et al. (1973).

Deviations from the regional west to east flow pattern may occur at the tributary buried bedrock valley to the south of the site (Figure E3-2). The discussion in Section E3.2 indicates the likelihood of flow from the Drift-Platteville aquifer to the St. Peter via the buried valley. The rate of this flow and its importance to horizontal movement in the St. Peter aquifer is unknown, however. Potentiometric surface measurements presented by Hult and Schoenberg (1981) indicate a head gradient from the buried valley towards well W23. This gradient may be the result of flow into W23, a multi-aquifer well which hydraulically connected the St. Peter to the lower head in the Prairie du Chien-Jordan aquifer prior to 1979. Additional multi-aquifer wells are found immediately east of the area in question and may influence the gradient as well. Also contributing to the northerly flow is the pumping well to the north of well W23 at well SLP3. In summary, although there is clearly a northward gradient component in the vicinity of the site, field information does not define the relative contributions to this gradient made by the buried bedrock valley, W23 and other multi-aquifer wells, or SLP3. And, despite the presence of a northward gradient component, the generally eastward gradient which characterizes the St. Peter aquifer must also be reckoned to determine the total resultant gradient. Flow from the buried bedrock valley into the St. Peter will flow in the direction of steepest descending gradient--not necessarily to the north directly to W23. Computer simulations and water table elevation data show that except possibly immediately above the tributary bedrock valley, flow in the Drift-Platteville aquifer will be directed east and southeast by the regional gradient.

Computer simulations were performed to evaluate likely pathways of contaminant movement in the St. Peter aquifer. Included were the effects of well SLP3, several industrial wells, and the buried bedrock valley assuming the leakage value of 9.3×10^{-9} per second as given in Section E3.2. Results from this simulation are shown in Figure E2-15 (potentiometric surface) and in Figure E3-4 (flow).

In general, the flow predictions indicate travel to the eastern and southern river boundaries at velocities on the order of 1 to 2 feet per day. More subtle trends are apparent in Figure E3-5, which plots travel paths beginning at points situated north and south of the site. In this figure the influence of the tributary bedrock valley is apparent. Although flow is generally to the east, travel from starting points north of the valley tend to flow due east to the Mississippi River while those originating south of the valley travel

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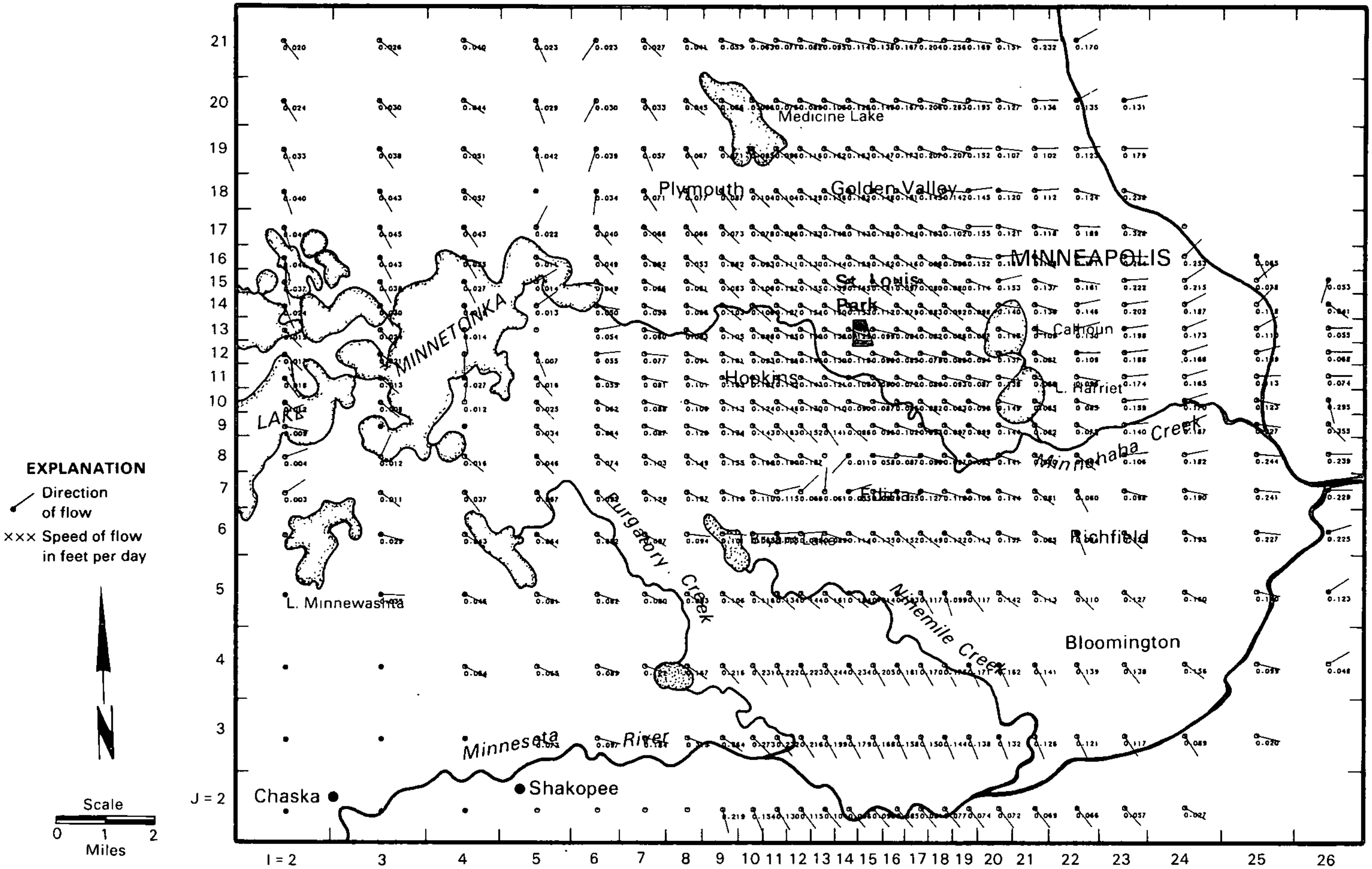

Figure E3-4 Pattern of Ground-Water Flow in the St. Peter Aquifer Predicted by Computer Model

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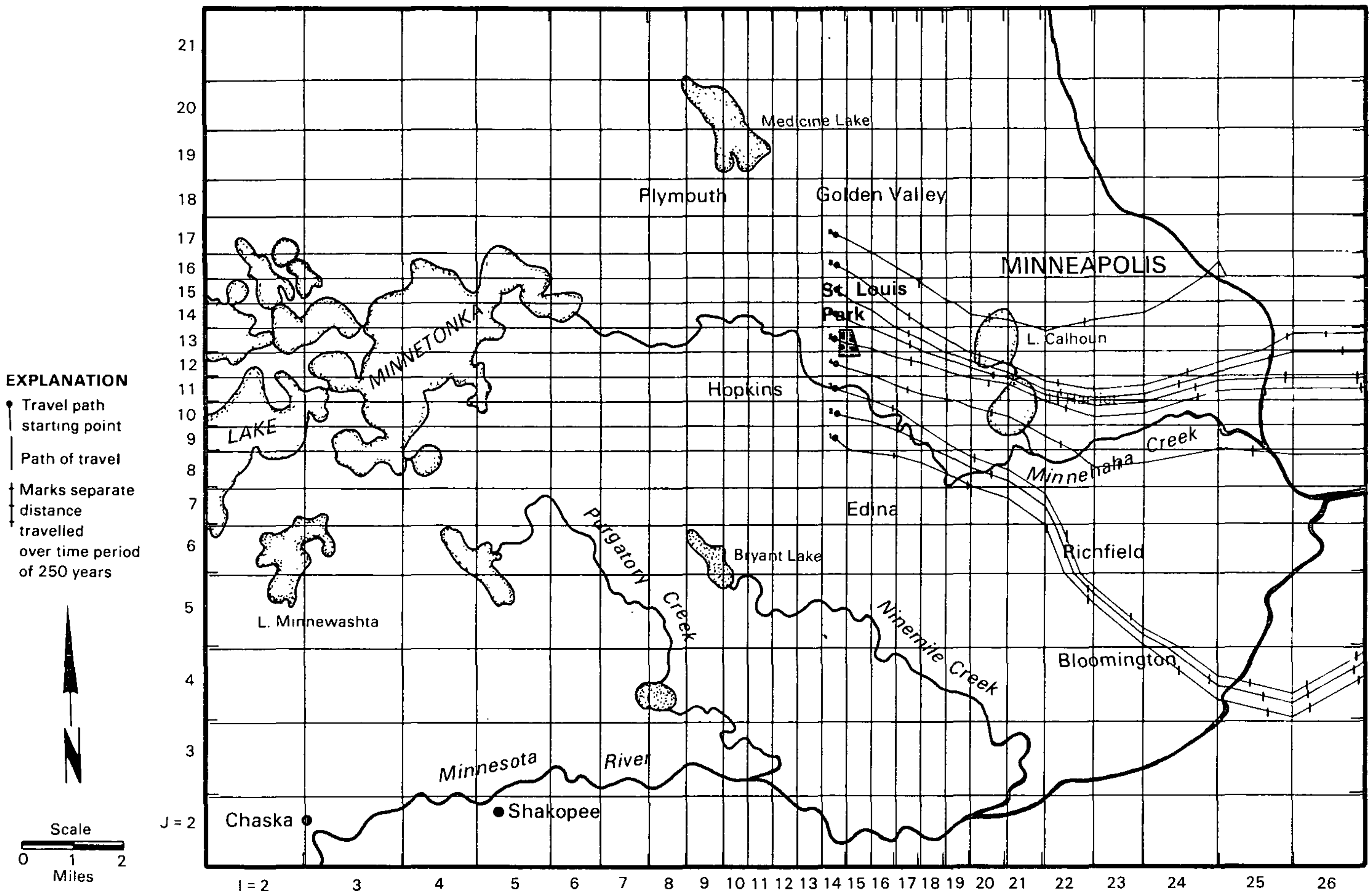

Figure E3-5 Selected Hydraulic Transport Travel Paths in the St. Peter Aquifer Predicted by Computer Model more to the southeast. This is most clearly seen at the final points of the travel paths. These computer simulations thus show the valley to act as a hydraulic barrier, dividing St. Peter ground-water flow into northern and southern destinations.

E3.3.2 Role of Buried Bedrock Valley

Figure E3-6 gives an approximate flow balance about the bedrock valley. This balance was determined from computer simulation results and shows the relative importance of vertical flow through the buried valley relative to the lateral flow intercepting the valley. The results are approximate, however they indicate the order of magnitude of the influence of the bedrock valley upon flow. As can be seen in Figure E3-6, as much as 30 per cent of the Drift-Platteville aquifer flow flowing above the bedrock valley may travel down to the St. Peter aquifer. The Drift-Platteville flow into the valley makes a substantial contribution to the flow in the St. Peter aquifer, accounting for over 60 per cent of the flow downgradient from the valley.

The Drift-Platteville flow which does not pass down the bedrock valley continues in the generally eastern and southern paths determined by regional gradients (Figure E3-2). To the east of the tributary bedrock valley is a major valley, presumably formed by an ancient channel of the Mississippi River. Within this area there is substantial hydraulic communication between the drift deposits which fill the valley and the St. Peter aquifer. In the center of the valley there is contact between the drift and Prairie du Chien Group. Contamination via these interfaces is not, however, a realistic concern. This is a consequence of the long travel times from the contamination source to these areas. Travel time to the St. Peter exposure is computed by the USGS model to be on the order of 50 years. Travel time to the Prairie du Chien is found to be approximately 6 decades. Over long travel times contaminant concentrations will be decreased by dispersion--a physical process in which contaminant is diluted by mixing with adjacent uncontaminated water. Concentration is also reduced by retardation, the term applied

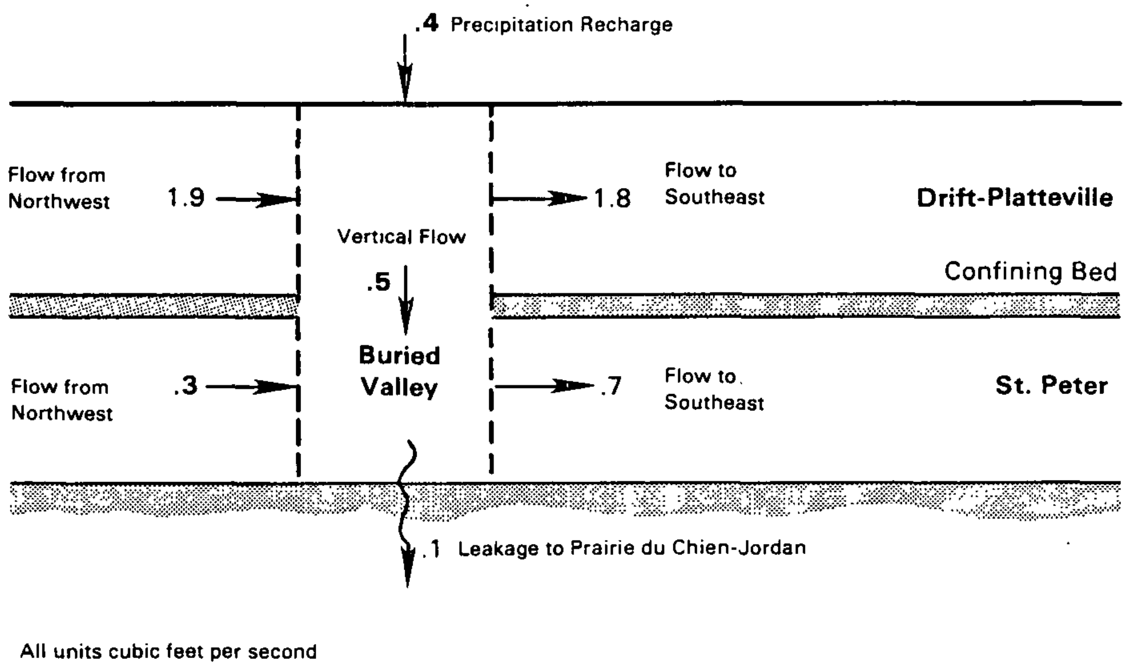

Figure E3-6 Approximate Water Balance for Tributary Buried Bedrock Valley as Determined from Computer Model Results

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to all chemical and physical processes which tend to remove contaminant from the water phase. Dispersion is primarily determined by the characteristics of the aquifer medium while retardation is a function of both the medium and the contaminant. If the retardation value for the Drift determined in Section E2.7 is applied, travel times are lengthened to 1000 years to the St. Peter exposure and 1100 years to the Prairie du Chien.

E3.3.3 Role of Multi-Aquifer Wells

A second possible route of contamination from the Drift-Platteville aquifer to the St. Peter is a multi-aquifer well. Hult (1979) lists eighteen wells in the immediate vicinity of the former plant site which penetrate both the St. Peter and Platteville aquifers. Subsequent logging of selected wells nowhere indicated flow from the Platteville to the St. Peter (Hult and Schoenberg 1981). These results do not, of course, rule out such flow as a possibility elsewhere, but they tend to show that multi-aquifer flow may occur less frequently between the Platteville and St. Peter than between the St. Peter and Prairie du Chien where significant flows were logged.

Previous investigators have implicated multi-aquifer wells as potential conduits to carry contamination from the Drift-Platteville to the St. Peter aquifer and eventually to St. Louis Park well SLP3. Barr (1979) uses ground-water model results to predict significant entrainment of contaminants from multi-aquifer wells W27 (Terry Excavating) and W33 (Midco Register) into SLP3. However, Barr's analysis appears to be based upon the assumption that W27 and W33 are the only multi-aquifer wells impacting the area. Subsequent work by Hult (1979) and Hult and Schoenberg (1981) indicate a large number of multi-aquifer wells connecting the Drift-Platteville to the St. Peter and the St. Peter to the Prairie du Chien-Jordan. The wells identified by these and other sources to be known or probable multi-aquifer wells are mapped in Figure E3-7. Wells connecting the St. Peter to the Drift-Platteville will tend to raise the potentiometric surface of the St. Peter and cause flow down the well and into the St. Peter.

EXPLANATION

- ■ Wells to Prairie du Chien — Jordan or deeper aquifers
- ● Wells to St. Peter Aquifer

Scale

0 0.5 1 Mile

Figure E3-7 Location of Multi-Aquifer Wells in Proximity to the Site Area

Connecting wells to the Prairie du Chien-Jordan will have an opposite effect -- causing a lowering of potential in the St. Peter and leading to flow from the St. Peter to the Prairie du Chien-Jordan. These two types of wells in close proximity will thus tend to counteract each other's influence on the St. Peter. As can be seen in Figure E3-7, there is a concentration of both well types to the east of the site area. The prevalence of industrial and commercial land use is presumably responsible for this concentration of relatively deep wells.

The net effect of the multi-aquifer wells to the St. Peter is that, while they may serve as conduits of contaminant to the St. Peter, they are unlikely to create a significant potentiometric high in the St. Peter. This is due to the counteracting influence of Prairie du Chien-Jordan multi-aquifer wells.

Another influence south of SLP3 with the potential to create a potentiometric high and drive flow to the north is the tributary buried bedrock valley. However, computations made with the USGS flow model in this study show no tendency for the buried bedrock valley to create a sufficient potentiometric high to divert flow northward either. Those computations indicate that flow to SLP3 comes from the west-northwest (Figures E3-4 and E3-5). In a more severe test of the model, SLP3 was simulated to be operating at full, rather than average, capacity. Results from this simulation show a small tendency for flow into SLP3 from the plant site and the bog area south of the site. Thus sustained operation of SLP3 at full capacity may lead to very slow migration of contaminated ground water from the area to the south. Historically, contamination of SLP3 has not been observed; however, the travel time from the site is at least 50 years. With retardation accounted for, travel time from the site to SLP3 exceeds 1000 years. Thus, the possibility of future contamination in SLP3 is remote.

E3.4 Analysis of the Prairie du Chien-Jordan Aquifer

E3.4.1 Overview

The Prairie du Chien-Jordan aquifer is the most extensively utilized of the area's aquifers. The cities of St. Louis Park, Hopkins, Edina and other neighboring municipalities draw substantial quantities of drinking water supply from this aquifer and a number of large industrial wells are screened in the Prairie du Chien Group and Jordan Sandstone as well. As a result of the numerous wells in the aquifer, there is a fairly complete network for observation of water quality and potentiometric surface elevation. These data are utilized in the discussion to follow as a touchstone to test the plausibility of the computer model results.

Computer simulations were performed to investigate the transport of contaminated ground water in response to pumping wells and recharge via multi-aquifer wells. Historical information indicates considerable variability in contaminant transport over time due to changing patterns in ground-water pumpage. The following picture emerges from the information compiled to date:

- On a regional scale, the flow of ground water in the Prairie du Chien-Jordan is from west to east, driven by a gradient of roughly 10 feet per mile.
- A concentration of multi-aquifer wells in south central St. Louis Park (to the south and east of the Republic Creosote site) creates a moderate potentiometric high due to leakage from the St. Peter aquifer. Locally, this high deflects ground-water flow from the regional pattern.
- Major pumping wells including St. Louis Park wells SLP5, SLP6, SLP8, SLP14, SLP15 and SLP16, Hopkins wells H1, H4 and H5, Edina wells E2, E6, E11, E13 and E16, and Methodist Hospital (W48) influence flow within their localities and on a regional scale. On the regional scale, the concentration of so many major wells within and adjacent to St. Louis Park creates a major sink area which intercepts much of the regional flow to the east.

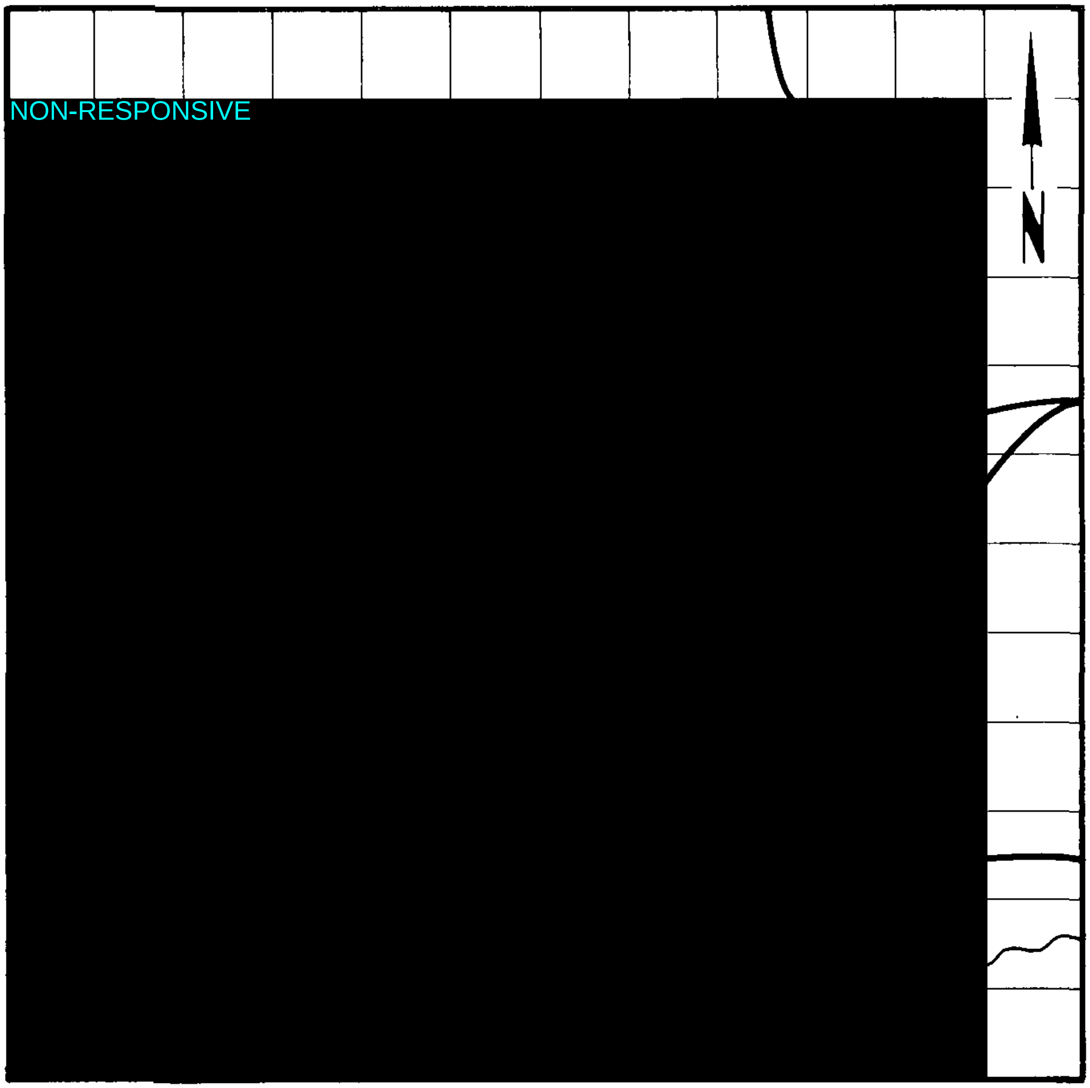
- Transport has varied over time in response to the abandonment of major muncipal supply wells and, on a shorter time scale, due to seasonal and other variations in well operating schedules.

Information to support these conclusions is presented in the remainder of this section.

E3.4.2 Transport from the Site

The recent history of PAH contamination in the Prairie du Chien-Jordan aquifer is summarized in Figures E3-8 and E3-9 with supplemental information given in Table E3-1. Appendix J discusses the data from which Table E3-1 is derived. The source of contaminant in the aquifer has yet to be fully explained. Most prior investigators presume well W23 on the plant site to be the primary source of PAH compounds in the Prairie du Chien-Jordan. The old sugar beet refinery well, W105, is another potential source, and there may be other deep multi-aquifer wells on or near the site. Earlier contamination associated with phenolic compounds may, for example, have reached the aquifer by W105 having been originally generated by sugar beet refining operations. Whatever the assumed source, analysis of transport using ground-water models shows that contaminants have been transported in all directions from the site over time. The results of the analysis are presented in the following paragraphs.

In the 1970's, a complex pattern of ground-water flow was produced by the many new major supply wells located about the site in all directions. Appendix G tabulates the various municipal wells in the St. Louis Park area and gives the dates of installation. Sunde (1974) hypothesized direct flow from site wells to neighboring wells depending upon which well was being pumped most heavily at any given time. Ground-water modeling studies by Barr (1977) did not bear this out however, showing that regional gradients to the east were not significantly interrupted by the municipal pumping centers. Thus, as of 1977, contamination of wells upgradient of the site in the regional flow path was unexplained.



EXPLANATION

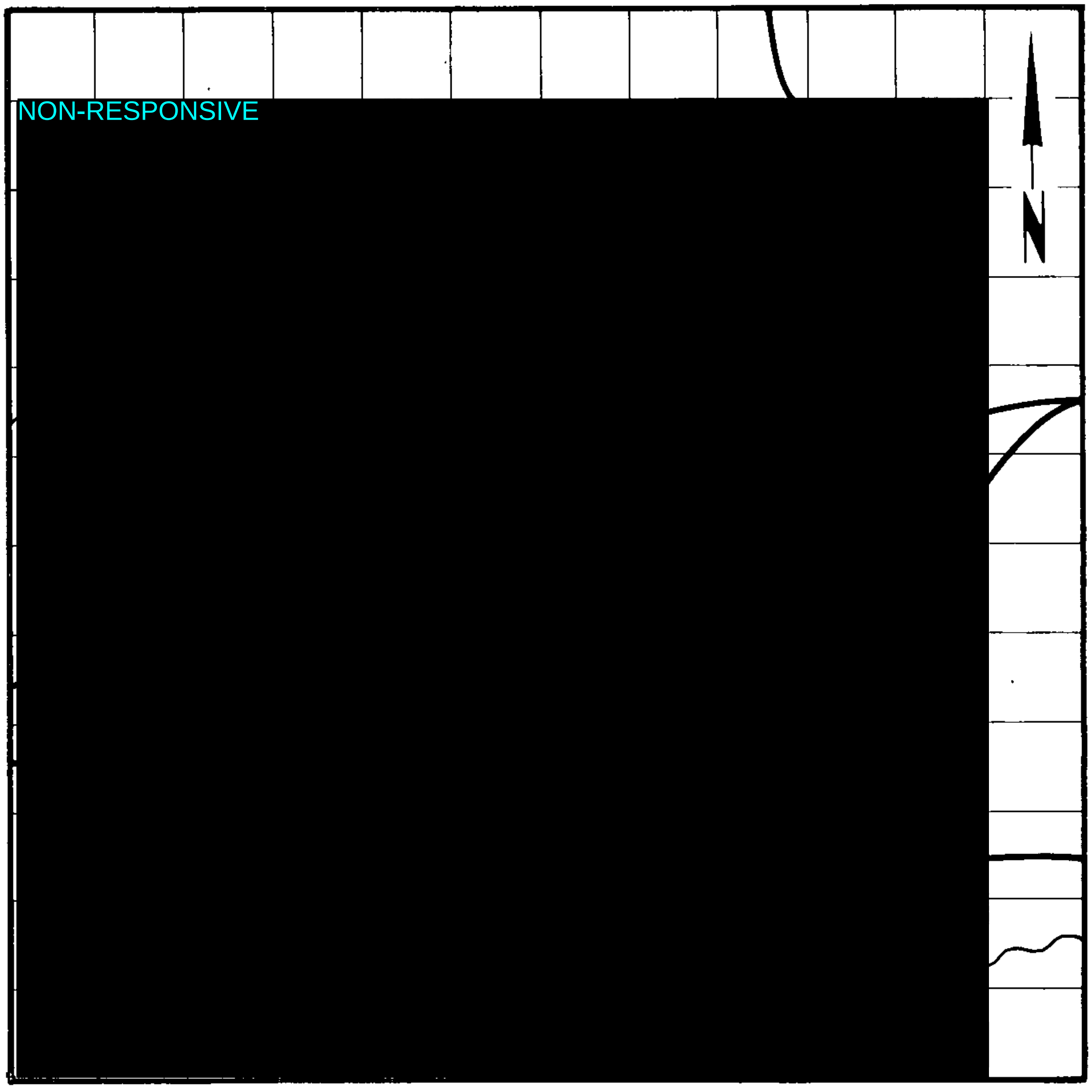
Concentrations of total PAH
in nanograms per liter

Data from Table E3-1

Scale

0 0.5 1 Mile

Figure E3-8 Measured Concentrations of Total PAH - Average for 1978-1979



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EXPLANATION

Concentration of total PAH
in nanograms per liter
Data from Table E3-1

Scale
0 0.5 1 Mile

Figure E3-9 Measured Concentrations of Total PAH - Average for 1980-1981

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TABLE E3-1

MEASURED CONCENTRATION OF TOTAL PAH

IN THE PRAIRIE DU CHIEN-JORDAN AQUIFER

Well Code	Concentration of Total PAH* in Nanograms Per Liter					
	1978-1979 data			1980-1981 data		
	Average	Range	Number of Samples	Average	Range	Number of Samples
SLP4	251	5-392	8	214	2-602	41
SLP5	7	7	1	5,299	10-29,215	10
SLP6	0	0-0	2	12	0-59	12
SLP7	62	0-123	62	87	0-427	24
SLP8	0	0-0	2	10	0-29	4
SLP9	116	0-232	2	70	3-224	20
SLP10	1,448	696-2,229	3	3,483	3,413-3,552	2
SLP14	28	14-51	3	105	11-360	10
SLP15	3,628	1,140-7,110	9	4,166	6-25,711	11
SLP16	3	0-6	2	0	0-1	4
H1	15	15	1	15	0-57	6
H3	39	39	1	4,068	0-10,794	9
H4	295	295	1	7	0-17	6
H5	21	21	1	16	2-73	7
H6	6	6	1	4	1-10	7
E2	9	3-13	4	5	0-16	12
E3	7	1-12	2	22	0-44	2
E4	20	1-47	3	9	0-39	9
E5	-	-	0	1	1	1
E6	2	1-2	2	7	0-33	12
E7	6	0-11	2	5	0-12	3
E8	-	-	0	16	16	1
E11	10	9-10	2	1	0-2	2
E13	6	1-10	2	2	0-4	4
E14	-	-	0	0	0	1
E15	5	0-10	2	3	0-7	3
E16	6	6	1	1	0-2	2
E17	0	0	1	37	37	1
E18	-	-	0	0	0	1
M11	-	-	0	0	0	2
M13	-	-	0	0	0	1
M14	-	-	0	0	0	2
W29	25,650	8,650-42,650	2	-	-	0
W32	20,675	20,675	1	-	-	0
W34	56	56	1	-	-	0
W35	0	0	1	-	-	0
W38	-	-	0	20,700	20,700	1
W48	-	-	0	866	414-1,317	2
W50	299	299	1	-	-	0

TABLE E3-1 (Continued)

Well Code	Concentration of Total PAH* in Nanograms Per Liter					
	1978-1979 data			1980-1981 data		
	Average	Range	Number of Samples	Average	Range	Number of Samples
W70	5,994	5,994	1	-	-	0
W80	-	-	0	11	11	1
W112	-	-	0	83	83	1

*Total PAH determined as the sum of acenaphthylene, anthracene, benzo(a) anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluorene, fluoranthene, indeno(1,2,3 -cd)pyrene, phenanthrene and pyrene.

This dilemma was investigated in the present study using the confined aquifer analytical model (Section E2.6) to examine the potentiometric surface and flow in greater spatial detail than allowed by the grid size selected for the USGS model. Initial simulations with the model were similar to those by Barr (1977) in that the effects of multi-aquifer wells were not included. These simulations did not predict transport paths from the site to the contaminated wells (Figure E3-10). The conclusions from these simulations would thus be the same as those reached by Barr (1977): that contamination of municipal wells in the Prairie du Chien-Jordan could not be explained.

A second series of simulations considered the influence of multi-aquifer wells and illustrated their significance. A list of likely multi-aquifer wells (Table E3-2) was compiled from various sources, primarily Hult (1979) (See also Appendix D.). For lack of field information, flow down these wells from the St. Peter aquifer to the Prairie du Chien-Jordan was assumed to be 150 gallons per minute per well--the value determined from flow measurements taken in W23 (Hult and Schoenberg 1981). With the inclusion of multi-aquifer flow, model results reproduced the minor potentiometric high observed in the Prairie du Chien-Jordan by Hult (1979) and Hult and Schoenberg (1981). The predicted potentiometric surface has a local high at the plant site and to the east (Figure E3-11). This high acts to deflect the regional flow pattern more to the south. It also acts as a sufficient barrier that intensive pumping at nearby wells can locally reverse the regional gradients. The reversals lead to flow to the north towards wells SLP10 and SLP15 (Figure E3-12), south towards well H3 (Figure E3-13) and west towards well SLP5 (Figure E3-14) when these wells are pumped at full capacity.

The results presented in Figures E3-12 through E3-14 are based upon steady-state conditions; it remains to be demonstrated that such an assumption is realistic. The time to reach a steady-state is indicated by the aquifer response time. The response time for a confined aquifer may be defined as:

$$\tau = \frac{r^2 S}{4T}$$

NON-RESPONSIVE

[redacted]

SIMULATION CONDITIONS

Average pumping rates at wells
No multi-aquifer well flow

Scale
0 0.5 1 Mile

Figure E3-10 Predicted Potentiometric Surface in Prairie du Chien-Jordan Aquifer without Multi-Aquifer Well Influence

TABLE E3-2

MULTI-AQUIFER WELLS TO PRAIRIE DU CHIEN-JORDAN IN PROXIMITY TO THE SITE INCLUDED IN MODEL SIMULATIONS

Well No.	Identification	Source*
W23	Republic Creosote deep well	H,U
W29	Flame Industries (in minor use as of 1976)	H,M
W32	Texatonka Shopping Center	H
W34	Crib Diaper Service	H
W35	Burdick Grain Co.	M
W38	Milwaukee Railroad Well	H
W40	Minnesota Rubber (in active use as of 1976)	H,M
W45	S-K Products, Inc.	H,M
W46	S-K Products, Inc. (in minor use as of 1976)	H,M
W47	Belco; Burdick Grain Co.	U
W49	Strom Black Co.	H,M
W50	Prestolite	U
W62	McCourtney Plastics (in active use as of 1976)	H,M
W66	Black Top Service	H
W69	Hedberg-Friedheim; Wolfe Lake	H,U
W70	Park Theatre	H
W74	Landers Gravel	S
W105	Minnesota Sugar Beet	H
W107	Interior Elevator Co.	H
W112	Old St. Louis Park Well No.1	H
W114	Hedberg-Friedheim	U

*Sources: M - Minnesota Department of Health high priority well list.
S - Sunde (1974)
H - Hult (1979)
U - Hult and Schoenberg (1981)

NON-RESPONSIVE

[redacted]

SIMULATION CONDITIONS

Average pumping rates at wells
Flow (150 gpm) in multi-aquifer wells

Scale
0 0.5 1 Mile

Figure E3-11 Predicted Potentiometric Surface in Prairie du Chien-Jordan Aquifer with Multi-Aquifer Well Influence

NON-RESPONSIVE

[redacted]

SIMULATION CONDITIONS

SLP7 and SLP15 full capacity pumping rates
Average pumping rates at all other wells
Flow (150 gpm) in multi-aquifer wells

Scale

0 0.5 1 Mile

Figure E3-12 Predicted Potentiometric Surface in Prairie du Chien-Jordan Aquifer when SLP7 and SLP15 Pump at Full Capacity

NON-RESPONSIVE

[redacted]

SIMULATION CONDITIONS

H3 full capacity pumping rate
Average pumping rates at all other wells
Flow (150 gpm) in multi-aquifer wells

Scale
0 0.5 1 Mile

Figure E3-13 Predicted Potentiometric Surface in Prairie du Chien-Jordan Aquifer when Hopkins Well 3 Pumps at Full Capacity

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NON-RESPONSIVE

[redacted]

SIMULATION CONDITIONS

SLP5 full capacity pumping rate
Average pumping rates at all other wells
Flow (150 gpm) in multi-aquifer wells

Scale
0 0.5 1 Mile

Figure E3-14 Predicted Potentiometric Surface in Prairie du Chien-Jordan Aquifer when SLP5 Pumps at Full Capacity

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where,

τ is the aquifer response time (T),

r is a specified distance at which reponse is to be evaluated (L),

S is the aquifer storativity, and

T is the aquifer transmissivity (L^2/T).

The response time gives the order of magnitude of time required for a change in the aquifer (due to a well starting to pump, for example) to be felt at a distance r. The wells of concern are within 1.5 miles of W23. With $S = 1 \times 10^{-4}$ (Reeder et al. 1976) and T = 56000 gallons per day per foot for the Prairie du Chien-Jordan, the response time is less than one day. It is reasonable to expect neighboring wells to operate at full or near-full capacity for one or several days. Thus, the predictions given in Figures E3-12 to E3-14 are physically meaningful.

The computer results do not, however, indicate the site as a source of contaminant for St. Louis Park wells 7 and 9, the PAH traces observed in wells SLP8, SLP14, H1, H5, nor the traces in Edina wells to the south. This same conclusion is reached by Sunde (1974) with respect to phenolics who suggested other sources or testing inaccuracies as the cause of these anomalies. Other sources of contamination remain a very real possibility. Further investigation is required to test this hypothesis.

E3.4.3 Regional Ground-Water Flow

The discussion above has focused on ground-water flow within the immediate vicinity of the former plant site. Another important consideration is the larger scale transport within the Twin Cities basin. It is this scale of transport that will determine the long-term migration of contaminants and thus their eventual destination.

Previous investigators have, for the most part, emphasized the regional gradients which cause transport from the western recharge areas to the Mississippi River. While this overall pattern is undeniable, there is an important deviation from this pattern in the St. Louis Park area. This deviation is the transport induced by the concentration of major pumping wells in St. Louis Park, Edina and Hopkins. Results from the USGS flow model have been used to determine the direction of mass transport in the Prairie du Chien-Jordan aquifer (Figure E3-14 and Figure E3-15). The results indicate that the municipal and other major wells act to intercept a significant portion of the regional flow. Consequently, there is a well-defined potentiometric low of large area roughly centered about St. Louis Park. Bounding this low on the east and south is a ground-water divide. Flow within the divide travels to the pumping wells while flow south or east of the divide continues to migrate south and east. Significantly, all wells found to have been contaminated by PAH compounds are located within this ground-water divide. Thus, it is very unlikely that PAH contamination within the Prairie du Chien-Jordan, if left to migrate, would in fact migrate far. For example, wells in Edina to the south of the divide are not in risk of contamination due to PAH transport within the Prairie du Chien-Jordan aquifer: the ground-water divide acts as a hydraulic "fence" separating these wells from contaminated areas so long as the pumping center continues.

The findings above are in concurrence with those of Sunde (1974, page 26). Sunde states: "Any contaminants entering the system should eventually flow to one of the municipal wells surrounding the site." Subsequent investigators narrowed the scope of their studies to too small an area to appreciate this fact. Modeling studies by Barr (1977) and Hickok (1981), for example, addressed an area entirely within the bounds of the ground-water divide.

E3.4.4 Contaminant Migration

One possible scenario for the future of contamination in the Prairie du Chien-Jordan is that PAH compounds will continue to travel with ground-water flow to eventually contaminate additional wells and necessitate their closing. The character of such a scenario was investigated with the USGS flow model based on the average well pumping rates given in Table E2-2. The model predictions are that ground-water originating at the site would flow southeast, reaching well SLP6 in roughly 65 years. If well SLP6 were then closed, flow would continue southeast and reach well E2 in another 30 years (95 years total travel-time). With well E2 closed, travel would continue, reaching well E3 in roughly 120 years total travel time and well E18 after over 200 years total travel time from the site. This analysis is overly pessimistic, in that contaminant will travel more slowly than water due to retardation and dispersion. Note, for example, that SLP6 is not now contaminated. Even if it is assumed that SLP6 will become contaminated in the very near future, this conservative analysis limits the number of wells which may become contaminated in the next 100 years to E2, E3 and possibly E4 and E6.

The analysis above points out a potential liability in well closure as a response to contamination. Particularly with high volume wells SLP6 and E2 operating, ground water travelling from the site is captured and removed by pumping. This is a part of the hydraulic "fence" effect described above. However, if these wells are closed, the hydraulic barrier is removed and contaminant is free to travel further to the southeast. However, hydraulic travel times in this case are still very long, for example, over 100 years to well E18 from well E2.

The problem addressed above relates to the hypothesis that a well-defined plume of contaminant is traveling through the Prairie du Chien-Jordan, eventually to contaminate wells south and east of the site and jeopardize the water supplies of Edina and other communities. These concerns are unfounded on two counts. First is the presence of the hydraulic fence described above. Secondly, the history of contamination in the Prairie du Chien-Jordan has been one of dispersing contaminant, a delaying influence on contaminant travel alluded to earlier. The computer simulations presented in Section E3.4.2 explain a mechanism for the seemingly inexplicable transport of contaminant to the north, the west, the southwest, the southeast and the east despite regional gradients to the east-southeast. The most likely explanation--and one supported by the computer simulation--is that PAH compounds have been dispersed in all directions as nearby high-capacity wells have operated periodically over time. This phenomenon has acted over time to dilute and disperse contaminant.

Dispersion of contaminant, plus the containing effect of large municipal wells, will prevent the future contamination of wells in Edina and further southwest from the site so long as those wells continue to operate. With these existing mechanisms at work, new gradient control wells or interception wells would be ineffective and redundant. Hickok and Associates (1981) employ analytical models to propose the installation of gradient control wells at a location "downstream" of the source area to serve as interceptors of migrating contaminant. This proposal is so severely flawed in concept and design that it offers no assurance of success. The following errors invalidate the design premise and the models employed by Hickok:

1. The model assumes a confined aquifer with west to east artesian flow. In fact, vertical recharge via natural and induced leakage is a major source of water within the Prairie du Chien-Jordan, and must be accounted with artesian flow as the makeup source of pumped withdrawals.

2. Item 1 notwithstanding, flow in the Prairie du Chien-Jordan in the vicinity of Hickok's proposed gradient control wells is subject to numerous local influences which divert the regional west to east flow. Hickok's analysis fails to account for these influences which include the numerous pumping wells throughout the area.

3. The installation of a gradient control scheme is redundant with the de facto system created by the existing municipal wells. To be sucessful, impractically large pumping rates would be necessary for Hickok's scheme to effectively compete with the existing wells.

In summary, the more detailed analysis of flow in the Prairie du Chien-Jordan aquifer conducted in this study has shown the gradient control well system designed by Hickok (1981) to be impractical, unworkable and of uncertain reliability.

E3.5 Analysis of Deep Aquifers

Beneath the Prairie du Chien-Jordan are two deeper and less-used aquifers: the Ironton-Galesville and Mt. Simon-Hinckley. There are no major wells drawing from the Ironton-Galesville and virtually nothing is published of the head distribution and flow in this aquifer. The Ironton-Galesville has not been examined in this study because of its low utilization as a water supply resource and due to the lack of field data to serve as a check on computer model results.

E3.5.1 Transport in the Mt. Simon-Hinckley Aquifer

The Mt. Simon-Hinckley is utilized as a source of drinking water through a number of deep municipal wells in St. Louis Park (SLP11, SLP12 and SLP13) and Edina (E9, E10 and E12). The observed potentiometric surface of this aquifer shows a distinct low encompassing these wells (Figure E3-15). Due to its great depth, few industrial and private wells have been constructed to the Mt. Simon-Hinckley. Hult and Schoenberg (1981) list only three in the St. Louis Park area: W23, the deep well on the Republic Creosote site, W38, the Milwaukee Railroad well roughly 0.8 mile east of the site, and W105, the abandoned well from the sugar beet refinery formerly on the plant site. None of these wells operates at present.

Previous investigators have suggested that wells W23, W38 and W105 may act as conduits bringing PAH contamination to the Mt. Simon-Hinckley aquifer (Hult 1979 and Hickok 1981). Analytical results have shown W23 and W38 water to have had high PAH concentrations in the past, however the composition of these samples includes water from multiple aquifers and little or no water from the Mt. Simon-Hinckley. The extent of contamination in the Mt. Simon-Hinckley alone cannot be isolated from these samples. Prior

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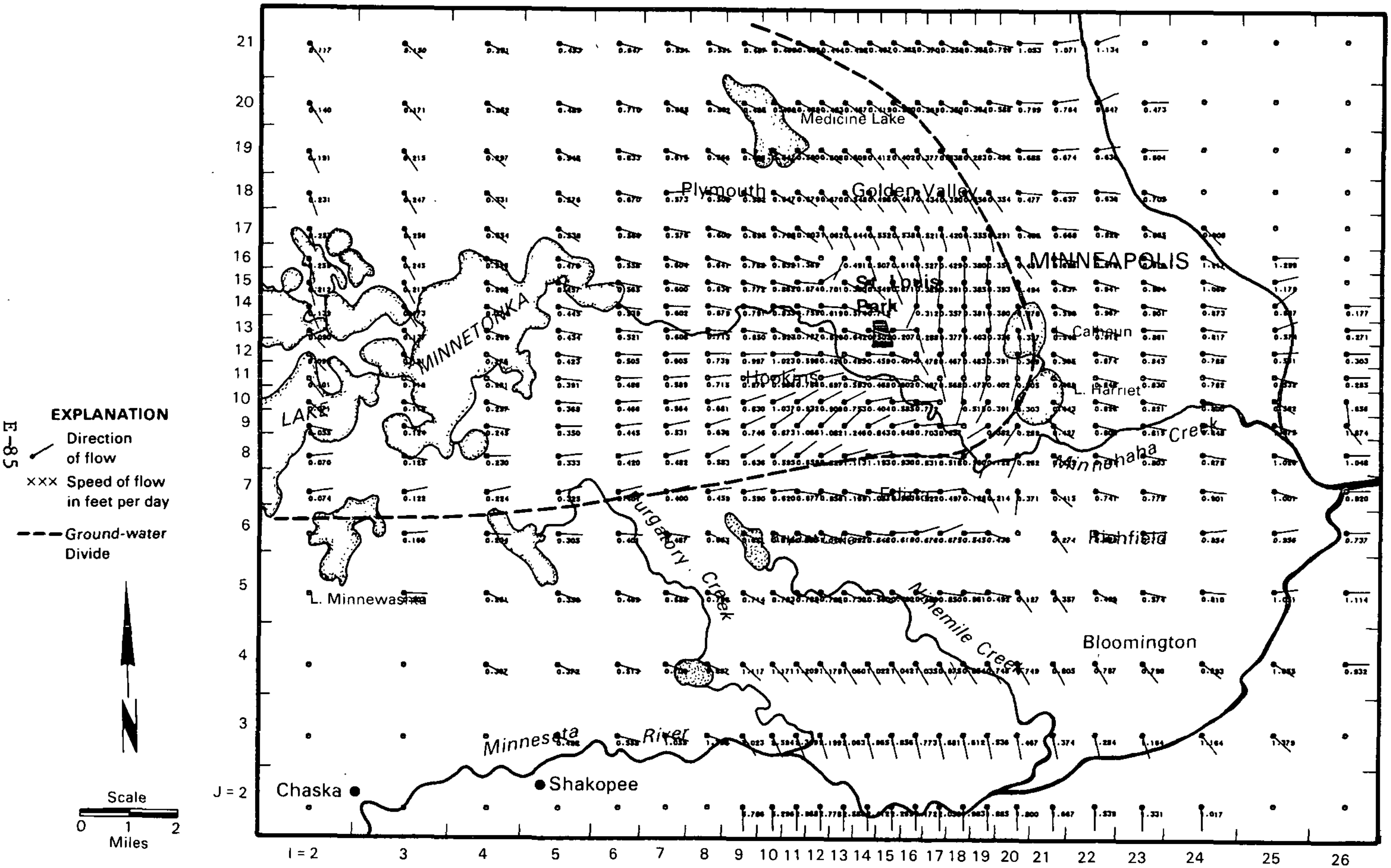

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Figure E3-15 Pattern of Ground-Water Flow in the Prairie du Chien-Jordan Aquifer Predicted by Computer Model to recent drilling activities, W23 was sealed below the St. Lawrence-Franconia confining bed and multi-aquifer flow to the Mt. Simon-Hinckley was small or none. W105 is reported to be filled with dirt, while W38 was at least partially obstructed until 1979 through much of the Mt. Simon-Hinckley. Thus, multi-aquifer flow is likely to have been quite low in recent years. Further, as shown in Appendix J there has been no measurable contamination in Mt. Simon-Hinckley wells.

Computer simulations were employed to estimate contaminant pathways in the Mt. Simon-Hinckley. Wells W23, W38 and W105 are located within one mile of each other. Less than one mile to the north of the wells is SLP11, an active well which operates roughly 50 per cent of the time. Capacity of this well is 1200 gallons per minute. To the south is SLP12, run about 15 per cent of the time with a capacity of 1400 gallons per minute. Two miles southwest is E12 with a capacity of 1000 gallons per minute operating 10 per cent of the time. The percentages of operation are computed for the St. Louis Park wells from the annual pumpages for 1972 through 1976; the information for E12 is based on 1976 data only.

Ground-water movement in the Mt. Simon-Hinckley on a regional scale was found in the computer simulations to be dominated by flow to the well centers (Figure E3-16). Typical velocities are on the order of 0.1 foot per day. A significant finding is that the plant site is directly adjacent to the centermost part of the regional low. Thus, migration of any contaminant which may reach the Mt. Simon-Hinckley below the site will be limited to a small area. Results from the USGS flow model (Figure E3-17) indicate that under average pumping conditions, flow will be from W23, W38 and W105 to SLP11. During periods when SLP12 or E12 are run frequently, gradients may be altered to direct flow to those wells--particularly when SLP11 is not also operating. However, over time SLP11 is probably the primary recipient of flow from the abandoned multi-aquifer wells due to its closer proximity and more frequent operation. Travel times from the site (W23) to SLP11 are predicted to be 33 years without accounting for contaminant retardation. If retardation is assumed to be roughly the same in the Mt. Simon-Hinckley Sandstones as in the St. Peter

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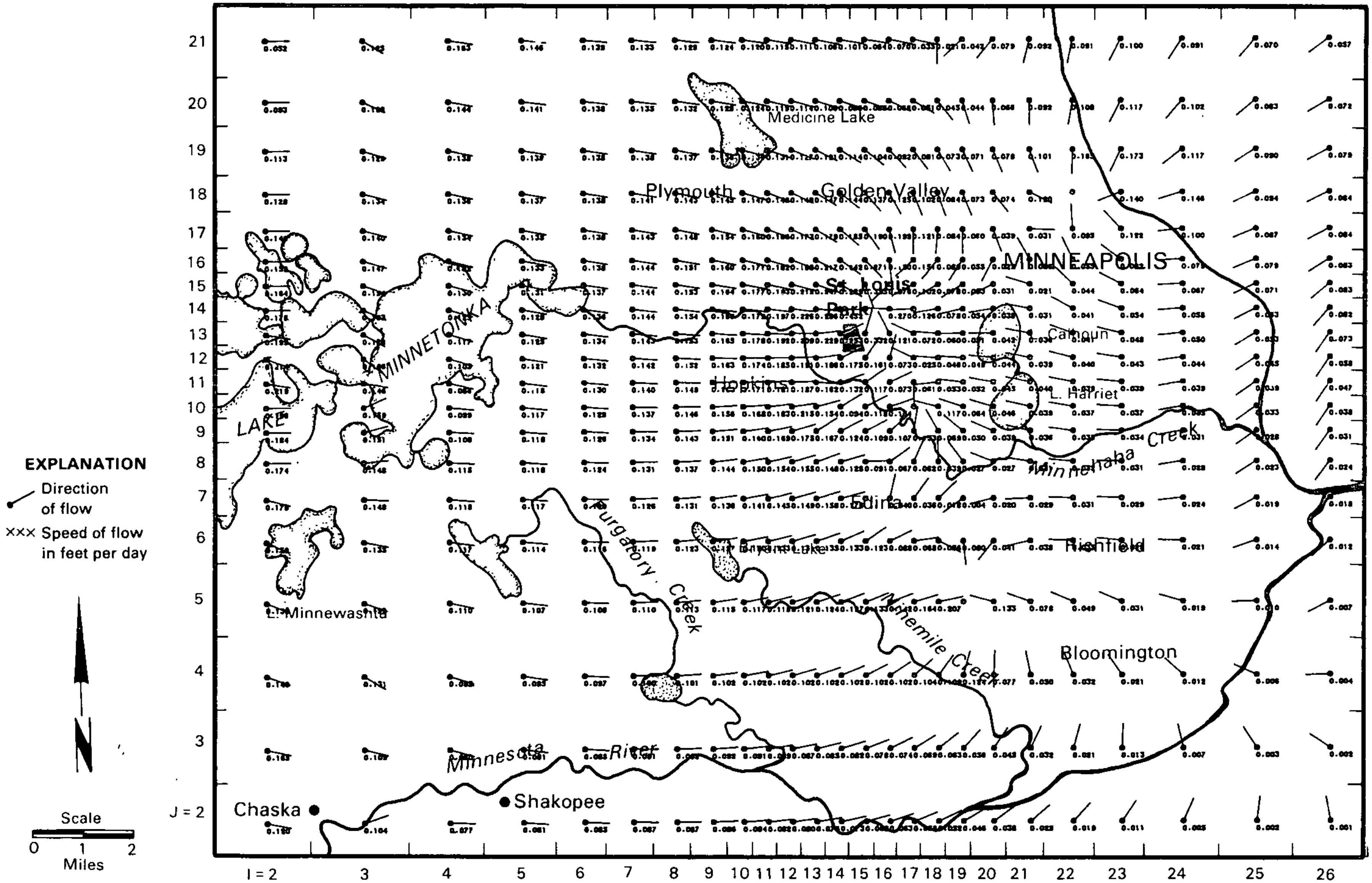

Figure E3-16 Pattern of Ground-Water Flow in the Mt. Simon-Hinckley Aquifer Predicted by Computer Model

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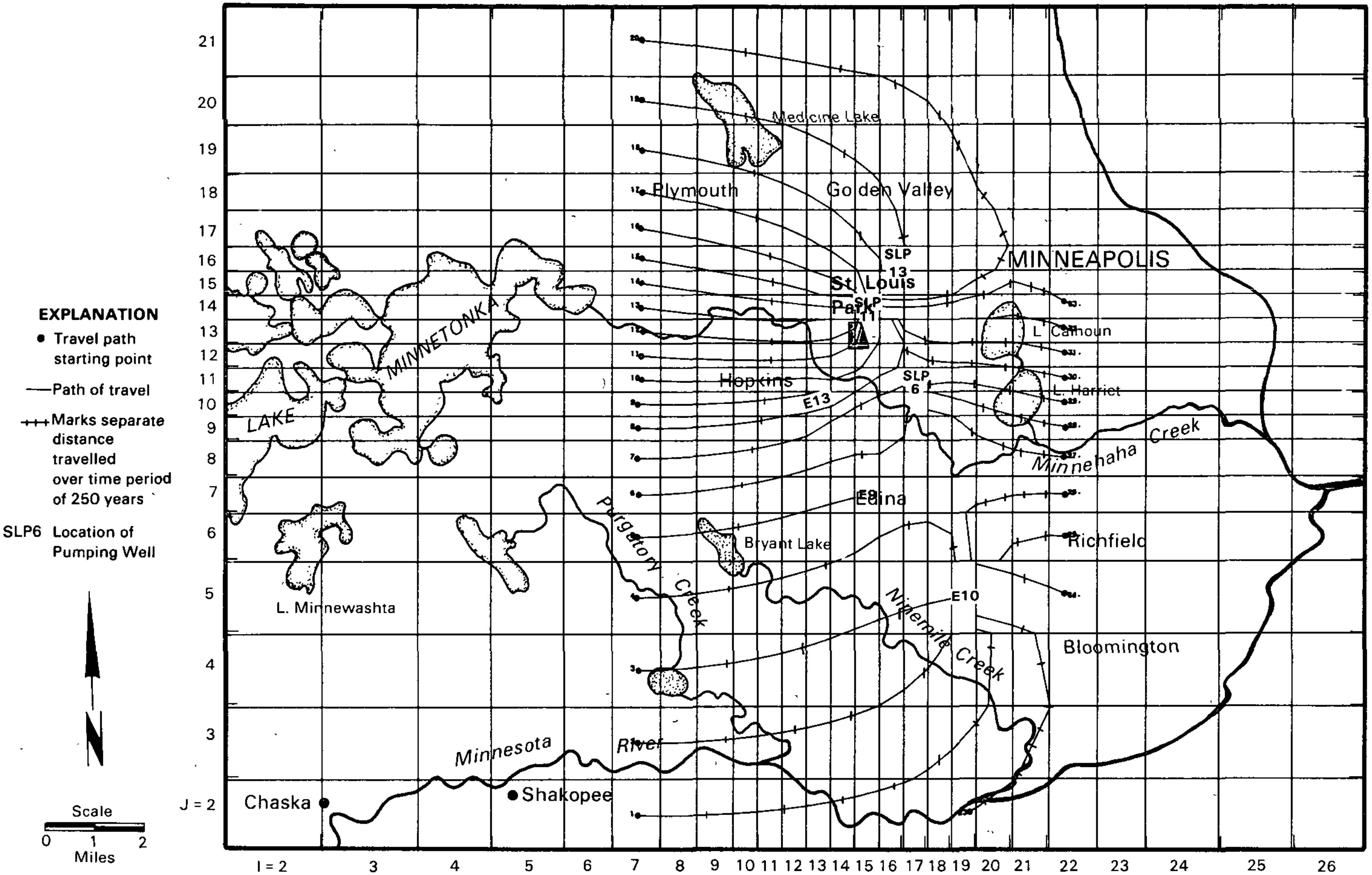

Figure E3-17 Selected Hydraulic Transport Travel Paths in the Mt. Simon-Hinckley Aquifer Predicted by Computer Model

Sandstone, then the travel time increases to over 600 years. Provided the present pattern of pumping remains essentially intact, it is unlikely that any contaminants which may reach the Mt. Simon-Hinckley aquifer below the site will migrate beyond the potentiometric low created by wells SLP11, SLP12 and E12.

E3.5.2 Future Development of the Mt. Simon-Hinckley

A possible solution to the problem of contamination in the Prairie du Chien-Jordan is to replace contaminated wells with deeper supply wells to the Mt. Simon-Hinckley. This is a remedy currently being pursued by the city of St. Louis Park in the installation of well SLP17 to the Mt. Simon-Hinckley in western St. Louis Park. However, the limits to this solution option are apparent in the present state of the Mt. Simon-Hinckley. As shown in Figure E2-13, a regional cone of depression, centered on St. Louis Park, already characterizes the potentiometric surface of the Mt. Simon-Hinckley aquifer. Only so much further drawdown of the aquifer is consistent with prudent management of this resource.

Norvitch et al. (1973) use the concept of available head to define the additional permissible pumping from the Mt. Simon-Hinckley. They define available head as that above the top of the aquifer. In other words, the top of aquifer is the practical limiting elevation to which the potentiometric surface may be drawn down.

The ground-water flow model was employed in a heuristic test of the Mt. Simon-Hinckley's ability to sustain additional pumpage. In this test the following wells which currently pump from the Prairie du Chien-Jordan were assumed to have been closed: SLP4, SLP5, SLP6, SLP7, SLP9, SLP10, SLP15, H3, E2, E3, E4, and E6. To replace this supply, five Mt. Simon-Hinckley wells, each with an average pumping rate of 940 gallons per minute (2.1 cubic feet per second), were assumed to be operating at the locations of wells SLP4, SLP5, H3, E2 and E4. This is considered a worst-case scenario for needed future well installations in the Mt. Simon-Hinckley.

For the conditions described above, the ground-water model predicts that the minimum available head, 60 feet, will occur at the location of Edina well E2. This is a small available head margin, and thus the scenario of Mt. Simon-Hinckley pumping stated above is roughly a maximum feasible pumping rate. Nevertheless, the simulation results illustrate that considerably greater quantities of water than pumped at present may be withdrawn from the Mt. Simon-Hinckley, and that replacement of present and future contaminated Prairie du Chien-Jordan wells by Mt. Simon-Hinckley wells appears to be technically feasible.

E4. CONCLUSIONS

The major features of the ground-water flow system determined by ground-water modeling include the following of concern to contaminant movement:

Drift-Platteville Contaminant in the Drift-Platteville at the site is unlikely to migrate to municipal supply well SLP3. Flow of water, and thus more slowly traveling contaminant, will tend to the east, driven by regional flow gradients. A portion of flow from the site is likely to reach a tributary bedrock valley southeast of the site and flow vertically through the valley and into the St. Peter aquifer. Hydraulic travel time from the site to the tributary bedrock valley is 10 to 20 years. Further to the east, a major bedrock valley creates a vertical connection between the Drift aquifer and the Prairie du Chien-Jordan aquifer. Hydraulic travel time from the site to this bedrock valley is roughly 60 to 70 years.

St. Peter Sustained pumpage of St. Louis Park well SLP3 is likely to draw ground-water from aquifer areas to the south, including the St. Peter aquifer beneath the site area. Otherwise, flow in the St. Peter will generally travel to the east following regional potentiometric gradients. Flow from the tributary bedrock valley will conform to this trend, flowing to the east towards the main bedrock valley where the St. Peter contacts the Drift and in turn the Prairie du Chien. Hydraulic transport travel time via the St. Peter is roughly 100 years from the tributary bedrock valley to the Prairie du Chien contact.

Prairie du Chien-Jordan The many large pumping wells in the Prairie du Chien-Jordan concentrated in St. Louis Park, Hopkins and Edina significantly influence flow in the site area. Long-term flow trends from beneath the site are to the pumping wells. Computer model results reveal a regional potentiometric low created by the concentration of pumping wells. The low is centered roughly beneath St. Louis Park and is bounded by a ground-water divide. The divide acts as a hydraulic control, preventing flow and thus contaminant migration beyond it to the south and east. Because all contaminated areas in the aquifer are within the divide, continuance of current pumping patterns will prevent contaminant from spreading further than St. Louis Park well SLP6 and northern Edina wells E2 and E3 with the unlikely exceptions of E4 and E6.

Local modifications of the large scale flow pattern described above are created during intervals when large capacity municipal wells are pumped. Such pumping creates temporary flow counter to the regional gradient. Ground-water modeling results indicate that ground-water in the Prairie du Chien-Jordan beneath the site may be drawn to nearby wells SLP5, SLP10, SLP15, and H3 when those wells operate at full pumping rate. SLP7 and SLP9 do not draw water from as far south as the site, however.

Mt. Simon-Hinckley Flow in the Mount Simon-Hinckley aquifer is strongly dominated by the effects of a few large municipal wells. These wells create a constant and extensive cone of depression which draws flow from all directions. Because of this effect, any contaminant reaching the Mt. Simon-Hinckley beneath the site is unlikely to travel beyond St. Louis Park wells SLP11 and SLP12 and Edina well E12 so long as they continue to pump. The geat majority of flow from beneath the site area will travel to SLP11.

Additional development of the Mt. Simon-Hinckley for water supply is possible. The capacity for development appears adequate to support alternative water supply proposals advanced in Chapter 6.

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APPENDIX F

WASTEWATER TREATMENT AND DISPOSAL ALTERNATIVES

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F1. INTRODUCTION

The following sections address the alternatives for treating and/or disposal of water from the activities suggested in previous sections:

- use Well W23 as a gradient-control well for the bedrock aquifers (see Appendix D);
- install gradient-control wells (2 wells with a combined pumping rate of 25 - 100 gallons per minute) for the shallow aquifers (see Appendix C); and
- start closed St. Louis Park water supply wells, if they are not to be used for drinking water supply (see Chapter 6).

The treatment and disposal options for these activities were evaluated based on the water quality of the aquifers, the proposed pumping rate for gradient control, and the feasibility and costs of treatment and disposal, including two on-site treatment alternatives and discharge to the Metropolitan Waste Control Commission (MWCC) plant via the St. Louis Park sanitary sewer system. For the purposes of treatment requirements and disposal options, the water quality characteristics of the ground water have been based upon the available data from previous reports. The data have been presented in Appendix K and summarized in Appendices B and C.

The disposal options for the water from both the Middle Drift and the bedrock aquifers have also been evaluated with respect to the impact (i.e., PAH concentration and flow) to the receiving water. In the case of pumping out large quantities of water (e.g., starting to pump all closed St. Louis Park wells and discharging instead of using the water), the volume of water determines the closest available receiving water. For the other scenarios the limiting factor in determining the acceptable receiving water is the water quality constraints and costs in meeting the discharge limitations.

TABLE F2-1

WASTEWATER QUALITY

	Total Organic Carbon (TOC) milligrams per liter	Phenols micrograms per liter	Total PAH micrograms per liter
Drift wells (25-100 gallons per minute)	5-20**	10-100**	1000-5000**
W23 (50-400 gallons per minute)	1-5*	2-10**	10-100***
St. Louis Park Wells			
5, 10, 15 (894 gallons per minute)	1-5*	2-10***	5-10***
4, 7, 9 (259 gallons per minute)	1-5*	2-10***	0.1-0.5***

*Estimate (see Appendices D and G).
**USGS 1980.
***Appendix J.

F2. WASTEWATER QUALITY AND QUANTITY

The following section summarizes the data available to characterize the wastewater to be disposed of from the three sources:

- Middle Drift gradient-control wells,
- W23 discharge, and
- St. Louis Park closed wells.

The following data are presented to define the treatment and disposal options available and compare costs associated with those options. The characteristics therefore include those constituents which affect the treatment and disposal of the water. The major concern is disposing of the wastewater containing PAH and related compounds. The ranges of expected organics are presented in Table F2-1 and are based on historical sampling data. Phenols are defined for the purpose of this analysis as measured by the 4AAP procedure.) The available PAH and phenols data are presented in Appendix K.

F2.1 Middle Drift Wastewater Characteristics

The expected concentrations of the water from the proposed gradient-control wells in the Middle Drift aquifer are based on existing data from the U.S. Geological Survey (USGS) monitoring wells. The wells which are considered to be representative of the water quality to be pumped from the new wells are USGS monitoring wells W9, W10, W11 and W16. The observed phenols and total PAH ranged from 10-100 micrograms per liter and 1000-5000 micrograms per liter respectively. The proposed location of the gradient-control wells for the Middle Drift are presented in Appendix C. The pumping rate was assumed to range from 25-100 gallons per minute and the quality is assumed to remain relatively constant throughout the pumping period.

F2.2 Prairie du Chien Water Quality

To estimate the quality of the water to be pumped from well W23, data for wells screened to the Prairie du Chien aquifer were summarized. The total PAH concentrations were found to be in the range of 10-100 micrograms per liter. The phenols concentrations were based on the USGS data for monitoring wells W29, W40, and W112. The phenols concentrations are approximately an order of magnitude less than the Middle Drift aquifer and TOC concentrations are estimated to be 1-5 milligrams per liter based on the lower phenol concentration. The range of pumping rates for W23 was from 50 to 400 gallons per minute and is expected to pump from the Prairie du Chien (see Appendix D).

The water quality for the St. Louis Park wells are based on sampling data from 1980-1982 for total PAH and phenol. The TOC concentrations are estimates based on CH2M Hill and Monsanto Research Co. MRC data (see Appendix G). The range of pumping rates for the St. Louis Park wells are based on historical, average pumping rates (see Appendix G).

F3. WASTEWATER TREATMENT/DISPOSAL ALTERNATIVES

The most cost-effective disposal of the water from gradient-control wells is dependent upon (1) the quality of water pumped, (2) the compatability of the wastewater to municipal treatment (both with respect to quality and quantity), (3) the nearest surface water body and (4) costs for the treatment and disposal.

The following section addresses the treatment/disposal options available for separate collection and treatment of the following sources:

- Middle Drift wells,
- Well W23, and
- St. Louis Park wells SLP4, SLP5, SLP7, SLP9, and SLP15.

The alternatives are considered separately although economies of scale could be realized in combining flows from the sources to be treated.

The ultimate disposal options include:

- Discharge to the St. Louis Park sanitary sewer system for treatment at the MWCC Pigs Eye Plant;
- Discharge to the St. Louis Park storm sewer system and subsequent discharge to the Minnehaha Creek or the Minneapolis Chain of Lakes; and
- Discharge to the Minneapolis storm sewer system and subsequent discharge to the Mississippi River.

F3.1 Middle Drift Wastewater Treatment and Disposal

The Middle Drift water is expected to be the most contaminated of the ground water which may be disposed of in the gradient-control scheme. The phenols and PAH and related compounds concentrations are expected to be 10-100 micrograms per liter and 1000-5000 micrograms per liter respectively. These concentrations are significantly greater than the expected allowable discharge to surface waters based on the St. Louis Park NPDES permit for storm sewer discharge (MPCA 1975) and the Minnesota Water Quality Standards (MPCA 1981). If the gradient control option is chosen the wastewater will have to be treated prior to the discharge to the Minnehaha Creek or Mississippi River. Discussion of the disposal options (direct and indirect discharge) are presented in the following section with any associated treatment requirements.

F3.1.1 Sanitary Sewer Discharge/Treatment at the Metropolitan Waste Control Commission Wastewater Treatment Plant

The major constituents in the Middle Drift wastewater are expected to be dissolved organics: phenols, cresols, and polynuclear aromatic hydrocarbons. These constituents are compatible with the activated sludge wastewater treatment provided at the Metropolitan Waste Control Commission plant. The plant has demonstrated adequate removal of the organics and PAH removals are reported to be >99 per cent (GCA 1982). Table F3-1 presents the available data for the MWCC plant removals. Influent concentrations for total PAH were reported to be 10-250 micrograms per liter and effluent concentrations were 0.1-10 micrograms per liter. The effluent concentrations are comparable to the upstream water quality of the Mississippi River (see Table F3-2). The current design capacity of the plant is 260 million gallons per day and the plant presently receives an average daily flow of 200 million gallons per day. (Gennere 1983). Therefore, there is adequate capacity for additional wastewater from Middle Drift gradient-control wells. The expected additional flow to the system is 36,000 to 144,000 gallons per day. In evaluating the capacity of the system, questions have arisen about the capacity of the interceptor sewer which will receive the flow. Since the Edgewood lift station will receive the flow there is adequate capacity for the flow. Approximately 1000 feet of 4-inch sewer would be required to connect to the 12-inch sewer along Lake Street as shown in Figure F3-1.

The Metropolitan Waste Control Commission has discharge limitations for seven metals, cyanide, oil and grease, pH and temperature. Table F3-3 presents the limits for discharges to the

HT177 8303002

NON-RESPONSIVE

[redacted]

Figure F3-1 St. Louis Park Sanitary Sewer System (Sewer Mains Near the Site)

TABLE F3-1

PAH DATA FOR THE METROPOLITAN WASTE CONTROL COMMISSION WASTEWATER TREATMENT PLANT*

All Concentrations are nanograms per liter	No. of Analyses	Influent Average	Standard Deviation	No. of Analyses	Effluent Average	Standard Deviation	Per Cent Removal
Naphthalene	7	1,800	2,400	8	ND	-	>99
Acenapthene	7	164	280	8	75	110	45
Fluorene	7	99,000	123,000	8	380	1,040	>99
Phenanthrene	7	22,000	27,000	8	32	89	>99
Fluoranthene	7	17,000	38,000	8	1,000	2,970	94
Anthracene	7	210	330	8	4.2	10	98
Triphenylene	7	860	1,370	8	ND	-	>99
Pyrene	7	15	40	8	1	3	93
Chrysene	7	110	200	8	1	1	>99
Benzo(k)fluoranthene	7	1,110	1,400	6	20	16	98
Benzo(b)fluoranthene	7	65	136	8	1.4	2.6	98
Benzo(a)pyrene	7	ND	-	8	ND	-	-
Benzo(g,h,i)perylene	7	310	420	8	1.4	1.6	>99
Dibenzo(a,h)anthracene	7	140	170	8	ND	-	>99
Indeno(1,2,3-cd)pyrene	7	ND	-	8	ND	-	-
TOTAL		205,800			1,500		>99

ND - Not Detected
*GCA 1982

TABLE F3-2

PAH DATA FOR THE MISSISSIPPI RIVER

(nanograms per liter)

		Sampling Location*			
	Monsanto[a]	UM859 GCA[b]	UM847 GCA[b]	UM840 GCA[b]	UM827 GCA[b]
Naphthalene	12	ND	20	ND	ND
Acenaphthene	NA	ND	10	ND	ND
Fluorene	0.96	340	600	20,200	ND
Phenanthrene	11	180	30	150	ND
Fluoranthene	4.8	ND	ND	ND	ND
Anthracene	<2	ND	2	2	2
Triphenylene	NA	ND	ND	ND	20
Pyrene	4.9	2	4	2	2
Chrysene	9.5	ND	1	2	3
Benzo(k)fluoranthene	<2	ND	ND	ND	30
Benzo(b)fluoranthene	<2	ND	ND	ND	10
Benzo(a)pyrene	<5	ND	ND	ND	1
Benzo(g,h,i)perylene	<8	ND	1	1	ND
Dibenz(a,h)anthracene	<5	ND	ND	ND	5
Indeno(1,2,3-c,d)pyrene	<11	ND	ND	ND	10
2-Methylnaphthalene	5.3	NA	NA	NA	NA
1-Methylnaphthalene	5.6	NA	NA	NA	NA
C_2-Naphthalene	17	NA	NA	NA	NA
Acenaphthylene	0.77	NA	NA	NA	NA
Benzo(a)anthracene	5.8	NA	NA	NA	NA
Benzo(e)pyrene	<5	NA	NA	NA	NA
Perylene	<5	NA	NA	NA	NA
Acridine	<10	NA	NA	NA	NA
TOTAL	133	520	670	20,400	80

*MWCC Wastewater treatment plant discharges between UM840 and UM827.
ND - Not Detected
NA - Not Analyzed
[a]Monsanto 1983., one analysis
[b]GCA 1982., average values
UM-Upper Mississippi Mile

TABLE F3-3

METROPOLITAN DISPOSAL SYSTEM
LIMITATIONS ON DISCHARGES

Substance or Characteristic	Limit
Cadmium, milligrams per liter	2.0
Chromium (total), milligrams per liter	8.0
Copper, milligrams per liter	6.0
Cyanide (total), milligrams per liter	4.0
Lead, milligrams per liter	1.0
Mercury, milligrams per liter	0.1
Nickel, milligrams per liter	6.0
Zinc, milligrams per liter	8.0
Temperature (except where higher temperatures are required by law), °F	150
pH, units	5.0-10.0
Oil and Grease, milligrams per liter	100

Metropolitan Waste Control Commission 1981.

MWCC treatment plant. In addition to these limits there is an annual strength charge for discharges containing suspended solids which exceed 275 milligrams per liter and COD which exceeds 580 milligrams per liter. None of the substances listed for control are expected to be present in quantities exceeding the limits or requiring an added strength charge (TSS and COD are expected to be < 100 and < 200 milligrams per liter respectively) and the wastewater is considered to be acceptable for discharge to the MWCC plant (MWCC 1981).

Since no pretreatment of the Middle Drift wastewater is required to meet the MWCC limits, the new pretreatment regulations for new and existing sources under the Petroleum Refining and Iron and Steel (cokemaking) categories were examined for comparison of treatment requirements prior to the discharge to a municipal system. For petroleum refining, the allowable phenols discharge range is 0.3-1.0 milligrams per liter and oil and grease is limited to less than 100 milligrams per liter. Pretreatment standards for byproduct cokemaking limit loadings for phenols (4AAP), benzene, naphthalene, and benzo(a)pyrene. (The phenol loadings are the average allowable BAT effluent concentrations. The toxic organics are the maximum allowable concentrations.)

phenols[a] (4AAP)	0.05-0.1 milligrams per liter
benzene[b]	0.05 milligrams per liter
naphthalene[b]	0.05 milligrams per liter
benzo(a)pyrene[b]	0.05 milligrams per liter

(a) Cokemaking, biological treatment (0.05 milligrams per liter) - physical/chemical treatment (0.10 milligrams per liter),

(b) no average concentrations have been established. The values represent the maximum allowable concentrations.

The Middle Drift wastewater concentrations are expected to be less than these standards. Since sufficient removal can be provided by the municipal wastewater treatment plant, no pretreatment is expected to be required.

F3.1.2 Discharge to the Minnehaha Creek

The storm sewers in the vicinity of the proposed Middle Drift gradient-control wells discharge to the Minnehaha Creek. An alternative to discharging to the MWCC plant is to provide on-site treatment and discharge to the Minnehaha Creek. Two major concerns in this alternative are the impact of the additional flow on the Creek and the water quality constraints for the discharge. Based on the St. Louis Park permit for the storm sewer and land farming discharge to the Minnehaha Creek, the effluent limits in Table F3-4 are assumed for the Middle Drift (MPCA 1975). The discharge volume in the permit is 173,900 gallons per day. Although the addition of the Middle Drift water to the system could double the discharge to the Minnehaha Creek during storm events, pumping could be minimized during those occurrances. The gradient-control wells could be pumped during dry periods and turned off during storm events. This procedure would augment the flow of the Minnehaha which has a seven day, ten year (7Q10) low flow of zero (MPCA 1975).

The wastewater effluent, however, would have to meet water quality standards because there would be zero dilution (see Table F3-4) during low flow occurrences. The water quality standard for phenols in the Minnehaha Creek is 10 micrograms per liter. Although no water quality criteria exist for total PAH, the MPCA has limited two indicator compounds for the St. Louis Park storm sewer discharge. (EPA is presently developing new criteria for individual PAH compounds and the 1980 criteria are not considered to be applicable criteria.) The limitations for the wastewater discharge to the Minnehaha would have to be negotiated during an NPDES permit aplication process. For the purposes of estimating costs for the wastewater disposal, the effluent limits are based on the water quality standard for phenols and the St. Louis Park storm sewer NPDES permit. Therefore, it is assumed that treatment will be required prior to the discharge. Effluent limits for phenols are expected to be 10-100 micrograms per liter with an associated removal of up to 90 per cent. The PAH limits are more uncertain since there are no criteria but are expected to be in the 1 to 10 micrograms per liter range (< 10 nanograms per liter

TABLE F3-4

EFFLUENT LIMITS[a] FOR ST. LOUIS PARK

STORM SEWER DISCHARGE TO MINNEHAHA CREEK

	Daily Maximum with Dilution[b]	Daily Maximum
Oil & Grease, milligrams per liter	0.5x	15
Phenols[c], milligrams per liter	0.01x	0.1
Quinone, milligrams per liter	0.04	0.4
Total Chlorine Residual, milligrams per liter	0.01x	0.2
Zinc, milligrams per liter	0.12x	1.0
Cadmium, milligrams per liter	0.03x	0.2
Copper, milligrams per liter	0.01x	0.5
Nickel, milligrams per liter	0.52x	2.0
Lead, milligrams per liter	0.03x	1.0
Ammonia, milligrams per liter	1.03x	2.0
Benzo(a)pyrene, micrograms per liter		0.01
Chrysene, micrograms per liter		0.01

[a] Minnesota Pollution Control Agency 1975.

[b] Concentration is adjusted for the Minnehaha Creek Flow:

$$x = \frac{[(0.25)(\text{flow in Minnehaha Creek}) + (\text{effluent flow})]}{[\text{effluent flowrate}]}$$

[c] Approved method for NPDES, phenols measurement is 4AAP which corresponds to ERT's definition of phenolics.

for carcinogenic compounds) with an associated removal rate of 99 per cent. It may be possible to obtain a permit for a larger allowable loadings of PAH and phenols.

To satisfy the effluent limits to permit discharge to the Minnehaha two technologies have been considered, carbon adsorption and ozone oxidation. Both systems should be capable of attaining 90 and 99 per cent removal of phenols and PAH respectively. The systems' design constraints and capabilities are presented in Section F4 with the costs for the alternatives.

F3.2 Discharge Alternatives for Well W23

Pumping rates for well W23 evaluated in this study ranged from 50 to 400 gallons per minute. The quality of the well water during the proposed period of pumping is expected to have the following characteristics (see Table F2-1):

- Total PAH — 10-100 micrograms per liter
- Phenols — 2-10 micrograms per liter
- TOC — 1-5 milligrams per liter

The limiting characteristics for discharge to the Minnehaha Creek or the Mississippi River are the PAH concentrations and the flow addition to the Minnehaha Creek. The phenols concentrations are within the water quality standards. To discharge the water, however, will require PAH removals on the order of of 90 per cent. The alternatives for this removal are discharging to the Metropolitan Waste Control Commission treatment plant with subsequent discharge to the Mississippi River or treating the waste prior to discharge to the Minnehaha Creek.

F3.2.1 Discharge to MWCC Plant

As previously discussed in Section F3.1.1, the MWCC wastewater treatment plant has demonstrated good removals of the PAH in the plant's influent. The PAH in the influent averaged 200 micrograms per liter in a recent sampling program (GCA 1982). The W23 wastewater is expected to have maximum concentrations in the same range. It is therefore assumed similar removals (> 99 per cent) will be achieved through the plant. With the high removal rate at the MWCC plant, the impact of W23 and Middle Drift gradient-control wells on the Mississippi River is not expected to be significant and no increase in the MWCC effluent concentration is anticipated. Figure F3-2 presents the expected additional loadings to the MWCC plant from the Middle Drift gradient-control wells and W23 and the loadings to the Mississippi River of PAH. The loadings for W23 and the Middle Drift aquifer are based on the expected ground-water concentrations presented in Table F2-1 at flows of 400 and 100 gallons per minute respectively. The Mississippi River and MWCC PAH concentrations are based on GCA data (see Tables F3-1 and F3-2) with average flows of 7,520 cubic feet per second (USGS 1980) and 200 million gallons per day (Genere 1983) respectively.

As previously discussed, the design capacity of the MWCC plant is adequate to handle the additional flow from both the Middle Drift and W23 at 50 to 500 gallons per minute. The combined flow adds approximately 0.2 to 0.7 million gallons per day to the 200 million gallons per day MWCC plant's average daily flow. The existing sanitary sewer system has a 12-inch main along Walker St. which connects to a 12- to 15-inch gravity sewer along Lake Street which flows northeast to Hampshire Avenue and then flows south to the St. Louis Park Lift Station #2 at the Edgewood Avenue (City of St. Louis Park 1977). This lift station has a pumping capacity of 5.6 million gallons per day (Barr 1977). The required sewer connection to the system (i.e., Walker Street sewer) will be approximately 500 feet of a 4- to 6-inch sewer.

F3.2.2 Discharge to Minnehaha Creek

Although the wastewater quality of W23 is expected to have less organics and PAH than the Middle Drift wastewater, the PAH are higher than the allowed discharge. Indicator PAH compounds, chrysene and benzo(a)pyrene, are limited to a maximum concentration of 10 micrograms per liter in the stormwater discharge to Minnehaha. It is

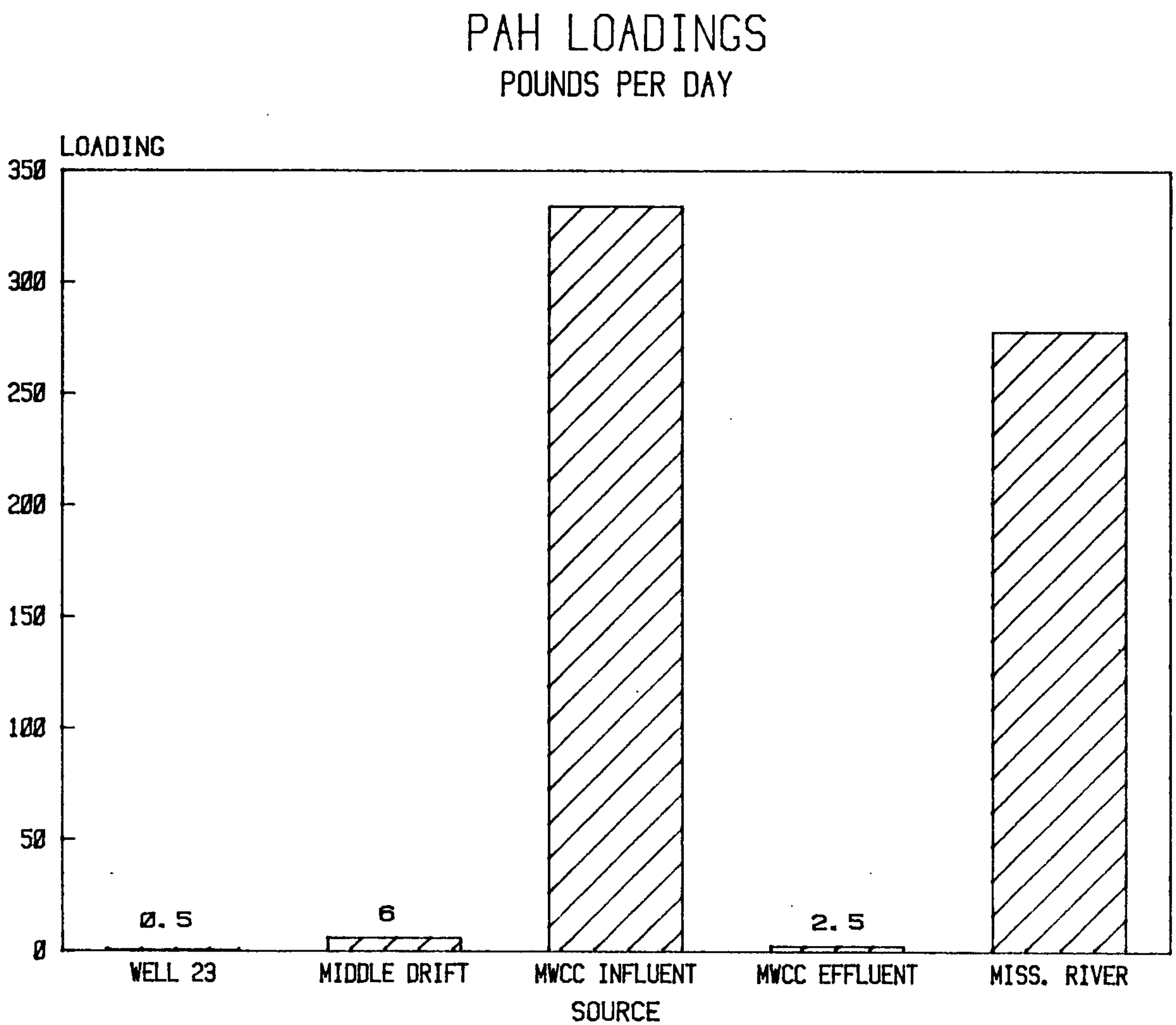

Figure F3-2 Comparative PAH Loadings to MWCC and Mississippi River expected a W23 discharge to the Creek would have a similar limitation and would, therefore, require treatment. Effluent concentrations for total PAH are expected to be in the range of 1 to 10 micrograms per liter assuming a 90 per cent removal efficiency through a treatment system. There are several treatment alternatives which can achieve that removal and are discussed in the following section F4.

There is an existing storm water treatment facility which treats surface drainage from the plant site and adjacent areas. To discharge treated wastewater to the Minnehaha Creek it is proposed to discharge to this system. Figure F3-3 presents the stormwater sewer system layout. Two lined storage ponds receive the runoff. The north pond has a storage volume of 23 acre-feet. The south pond has a volume of 44 acre-feet. The storm water flows by gravity from the north pond to the south pond. The lift station at the south pond, with two pumps (10,000 and 2500 gallons per minute capacities), intermittently discharges the storm water to the Minnehaha Creek.

A chlorination/dechlorination treatment system to oxidize organics prior to the discharge was installed but apparently has not been utilized to date. It was designed to oxidize 90 pounds of phenolics per day. Although this technology could be considered for treatment of Well W23 wastewater, recent studies by CH2M Hill have shown only selective oxidation by this method and undesirable byproduct compounds may be formed. Chlorination/dechlorination is therefore not considered for treatment. Further discussions of applicable treatment technologies are presented in section F4.

F3.3 Closed St. Louis Park Water Supply Wells

The closed St. Louis Park wells may be used in the future for water supply if adequately treated (see Appendix G). If not used as a water supply, it may be necessary for the wells to be pumped as gradient control wells and the water disposed of as wastewater. The flow from the combined wells would range from 900 to 1200 gallons per minute and would have total PAH concentrations in the range of 1 to 5 micrograms per liter which are concentrations acceptable for discharge to the Mississippi River or the Minneapolis Chain of Lakes. The

NON-RESPONSIVE

[redacted]

wastewater may not be discharged to the MWCC plant because the well water quality equals the MWCC effluent quality (PAH concentrations). This additional flow would be an undesirable hydraulic load on the system. The constraint for the discharge is therefore the volume, 900 to 1200 gallons per minute or 1.3 to 1.7 million gallons per day. This volume is not easily disposed of and may not be discharged to the Minnehaha Creek because it would significantly increase the flow (recorded flows are 0 to 15 cubic feet per second or 0 to 10 million gallons per day).

The closest discharge alternative is to discharge all wells except SLP5 to the Minneapolis Chain of Lakes. A new sewer main to the Minneapolis storm sewer system would be required for the option to discharge to the Mississippi. SLP5, which is expected to have a flow of less than 200 gallons per minute, is expected to have a small enough flow to be discharged to the Minnehaha Creek. The water level in the Minneapolis Lakes has declined since the mid-1900's to the extent it was necessary to pump water from the Mississippi River to maintain the water level in the lakes (Hickok 1969). The water is chlorinated prior to discharge to the lakes to reduce fecal coliform counts. Based on the information available on the Mississippi River PAH concentrations (see Table F3-3), it is assumed the water quality from the closed St. Louis Park wells would equal or be better than that from the Mississippi River. The pumping from the wells could also be scheduled to meet the needs of maintaining the water levels. The sewer requirements to discharge the closed St. Louis Park wells are:

	Sewer Connection	Distance
SLP7 and SLP9	Connect to West 26th Street	1000 feet
SLP15	Connect to Jersey Avenue Jersey Avenue to West 26th Street West 26th to France Avenue (to Cedar Lake)	1000 feet
SLP4	Connect to Natchez Avenue Natchez Avenue to Excellsior Ave. (to Lake Calhoun)	500 feet
SLP5	Connect to Wyoming Avenue	500 feet

The combined sewer requirements are 3,000 to 4,000 feet (the capacity of the existing lines are being investigated and sewers from West 20th Street to Cedar Lake and Excellsion Avenue to Lake Calhoun are not expected to be required). The above estimates are presented to estimate costs for the discharge which are presented in Section F5.

F4. AVAILABLE ON-SITE WASTEWATER TREATMENT TECHNOLOGIES

The on-site treatment alternatives for W23 and the two gradient-control wells for the Middle Drift aquifer are presented in the following discussion. For the purposes of this discussion it is assumed that if on-site treatment is necessary, all three of the wells would be treated at a single central facility. The combined flow to the plant is estimated to be 75-500 gallons per minute with PAH concentrations of 100-1000 micrograms per liter and TOC concentrations of 1-10 milligrams per liter.

The wastewater treatment technologies available in the treatment of ground water is limited to adsorption or oxidation of organics. The organic concentration in the wastewater is not expected to be sufficient to support biological treatment. The available wastewater treatment technologies are therefore equivalent to traditional water supply treatment technologies. Although the percentage removals are not as stringent as water treatment and the carbon and chemicals requirements are not as great; the technologies are the same.

F4.1 Granular Activated Carbon

Granular activated carbon (GAC) is a widely accepted method to remove organics and to provide potable drinking water supplies. Several studies have been performed to evaluate removal efficiencies, carbon requirements and contact times. Removals of 80 to greater than 99 per cent have been reported for GAC treatment of phenols and PAH in water supplies. Based on information from recent CH2M Hill (1982) studies, activated carbon can provide greater than 99 per cent removal of both carcinogenic and total PAH with influent concentrations comparable to SLP15 (total PAH of 6 to 7 micrograms per liter). Similar removals are expected for W23 and Middle Drift wastewaters to allow disposal to the Minnehaha Creek.

Studies have shown adsorption isotherms for several PAH compounds. These isotherms show adsorption capacities of 1.0 to 600 milligrams of organic compound per gram carbon. Recent studies however have been completed (CH2M Hill 1982) which have demonstrated capacities of 1.0 to 3.0 milligrams of total PAH per gram carbon. The capacity of the carbons are directly related to the influent concentrations of the PAH and phenols. A comparison of the carbon capacities (X/M) was completed to define the carbon requirements for the Middle Drift and W23 wastewaters. Table F4-1 presents the comparison.

The data available at PAH concentrations of 1 milligram per liter show carbon capacities of 300 to 700 milligrams compound per gram carbon. In using this data (500 milligram per gram) to estimate the carbon requirements for the Middle Drift and W23 wastewater, a carbon requirement of 5000 pounds per year is derived. The combined flow is expected to have PAH concentrations of 500 to 2500 micrograms per liter.

These carbon requirements are presented for cost estimating purposes only. Further information on the carbon requirements would be required through a pilot plant study prior to the final design of a treatment system.

F4.2 Ozone Treatment

Recent studies by CH2M Hill (1982) on St. Louis Park well SLP15 have included the oxidation of PAH compounds by ozonation with and without ultraviolet light. These studies showed good removal of PAH with and without UV radiation. Effluent concentrations of PAH and related compounds were less than 3 micrograms per liter for both 1 milligram per liter and 5 milligrams per liter ozone doses (and 20 minutes contact time). Carcinogenic PAH were less than 10 nanograms per liter. Although the expected PAH concentrations in the wastewater influent is expected to be greater than the SLP15 PAH concentrations found in the CH2M Hill studies, the removal efficiencies are expected to be comparable. Further studies are required to determine the ozone addition requirements and to identify compounds which may not be oxidized in the process.

TABLE F4-1

COMPARISON OF CARBON ISOTHERM DATA

(X/M, milligrams of compound removed per gram carbon)

	10 nanograms per liter	100 nanograms per liter	1,000 nanograms per liter	10 micrograms per liter	100 micrograms per liter	1,000 micrograms per liter
Fluoranthene	0.17[a]	0.3[a]		41	164	664
Anthracene	0.035[a]	0.06		15	75	376
Fluorene				89	170	330
Phenanthrene	0.025[a]			29	78	215
Acenaphthylene				21	49	115
Dibenzo(a,h)anthracene				2.1	12	69
Benzo(a)pyrene	0.004[b]			4.5	12	34
Benzo(g,h,i)perylene				2	4.6	11
Total PAH		0.4[c]	1.0[c]			
Phenol				1.7	6	21

[a]Calgon - 300 carbon, CH2M Hill 1982.
[b]Witco carbon, CH2M Hill 1982.
[c]All carbons, CH2M Hill 1982.
10 to 1,000 nanograms per liter data from CH2M Hill 1982.
10 to 1,000 micrograms per liter data from EPA 1980c.

F5. WASTEWATER TREATMENT/DISPOSAL COSTS

The following costs for W23 and the Middle Drift discharges are presented separately for the MWCC option and combined if on-site treatment is chosen. Tables F5-1 through F5-6 present the cost summaries for the alternatives for treatment and disposal of W23 and Middle Drift gradient-control wells, and St. Louis Park wells wastewater discharges. Based on the cost estimates, the present values of the treatment/disposal options were calculated (at 5 per cent interest) and graphically compared in Figure F5-1.

TABLE F5-1

MIDDLE DRIFT WASTEWATER MWCC DISCHARGE
TREATMENT/DISPOSAL COSTS

Capital Costs:	
Sewer to connect to St. Louis Park	
sanitary sewer, 1000 feet and street restoration	
$30.00 per linear foot	$30,000
Annual Costs:	
MWCC Sewer Charge 70¢ per 1000 gallons (MWCC 1982)	
at 100 gallons per minute - Annual Total	$37,000
at 25 gallons per minute - Annual Total	9,250

TABLE F5-2

WELL W23 WASTEWATER TREATMENT AND DISPOSAL COSTS

MWCC DISCHARGE

Capital Costs:	
Sewer Connection to St. Louis Park Sanitary Sewer/Street Restoration, 500 feet at $30.00 per linear foot	$ 15,000
Annual Costs:	
MWCC Sewer Charges at 70¢ per 1000 gallons	
50 gallons per minute - Annual Total	$ 18,400
400 gallons per minute - Annual Total	$147,000

TABLE F5-3

ON-SITE WASTEWATER TREATMENT AND DISPOSAL FOR WELL W23

AND MIDDLE DRIFT GRADIENT-CONTROL WELLS

Granular Activated Carbon Package Units
15 Minute Contact Time 20,000-40,000 lb Carbon

Capital Costs (100 gallons per minute):	
Package Unit (EPA, 1979)	$ 50,000
Building & Electrical	$ 20,000
Piping & Pumps	$ 5,000
Subtotal	$ 75,000
Contingency (25%)	$ 18,800
Subtotal	$ 93,800
Engineering & Studies (10%)	$ 9,400
Treatment Total	$103,200
Storm Sewer Connection 1000 linear feet	$ 30,000
Total Capital	$133,200
Operating and Maintenance (100 gallons per minute):	
Carbon Replacement $1.00/lb 5,000 lb	$ 5,000
Electricity, Labor & Maintenance (EPA, 1979)	$ 5,000
Annual Total	$ 10,000

Capital Costs (500 gallons per minute):	
2 package Units (EPA, 1979)	$200,000
Building & Electrical	$ 30,000
Piping & Pumps	$ 10,000
Subtotal	$240,000
Contingency (25%)	$ 60,000
Subtotal	$300,000
Engineering & Studies (10%)	$ 30,000
Treatment Total	$330,000
Storm Sewer Connection 1000 linear feet	$ 30,000
Total Capital	$360,000
Operating and Maintenance (500 gallons per minute):	
Carbon Replacement $1.00/lb 11,000 lb	$ 11,000
Electricity, Labor & Maintenance (EPA, 1979)	$ 14,000
Annual Total	$ 25,000

TABLE F5-4

ON-SITE WASTEWATER TREATMENT AND DISPOSAL FOR WELL W23

AND MIDDLE DRIFT GRADIENT-CONTROL WELLS

Ozone Oxidation 10-50 pounds per day Ozone
30 Minute Contact Time - No Ultraviolet Radiation

Capital Costs (100 gallons per minute):		
Ozone Generator (EPA, 1979)		$ 70,000
Contact Chamber (EPA, 1979)		$ 17,000
	Subtotal	$ 87,000
Contingency (25%)		$ 22,000
	Subtotal	$109,000
Engineering & Studies (10%)		$ 11,000
	Treatment Total	$120,000
Storm Sewer Connection		$ 30,000
	Total Capital	$150,000

Capital Costs (500 gallons per minute):		
Ozone Generation (EPA, 1979)		$210,000
Contact Chamber (EPA, 1979)		$ 25,000
	Subtotal	$235,000
Contingency (25%)		$ 59,000
	Subtotal	$294,000
Engineering & Studies		$ 29,000
	Treatment Total	$323,000
Storm Sewer Connection		$ 30,000
	Total Capital	$353,000

Operating and Maintenance (100 gallons per minute):	
Total O&M (EPA, 1979) Annual Total	$ 11,000

Operating and Maintenance (500 gallons per minute):	
Total O&M (EPA, 1979)	$ 25,000

TABLE F5-5

DISCHARGE SLP4, SLP7, SLP9, AND SLP15 TO MINNEAPOLIS
CHAIN OF LAKES AND SLP5 TO MINNEHAHA CREEK

Discharge to Minneapolis Chain of Lakes (SLP4, SLP7, SLP9, and SLP15) and Minnehaha Creek (SLP5)

- 2000 to 3,000 linear feet to Cedar Lake (SLP7, SLP9, SLP15)
- 500 linear feet to Lake Calhoun (SLP4)
- 500 linear feet to Minnehaha Creek (SLP5)

Capital:

• SLP7, SLP9, and SLP15 to Cedar Lake at $30.00 per linear feet, 2000-3000 linear feet	$60,000 to $ 90,000
• SLP4 to Lake Calhoun at $30.00 linear feet, 500 linear feet	$ 15,000
• SLP5 to Minnehaha Creek 500 linear feet	$ 15,000
Total	$90,000 to $120,000

Annual Operation and Maintenance:

5% per year (additional for storm sewer)	$ 7,000 to $ 10,000
Well pumping costs (200 feet head, 100-300 million gallons per year for each well, 5 wells)	$25,000 to $ 75,000

TABLE F5-6

PRESENT VALUE OF TREATMENT AND DISPOSAL ALTERNATIVES*

	MWCC	Onsite Treatment GAC	Onsite Treatment Ozone	Discharge to Minneapolis Chain of Lakes
Middle Drift				
25 gallons per minute	$220,000	-	-	-
100 gallons per minute	$770,000	-	-	-
Well W23				
50 gallons per minute	$380,000	-	-	-
400 gallons per minute	$2,955,000	-	-	-
Well W23 and Middle Drift				
100 gallons per minute	$766,000	$332,000	$368,000	-
500 gallons per minute	$3,706,000	$856,000	$849,000	-
St. Louis Park Wells used a gradient control	-	-	-	$725,000 to $810,000

*Assumes capital replacement after 30 years, 5% annual interest.

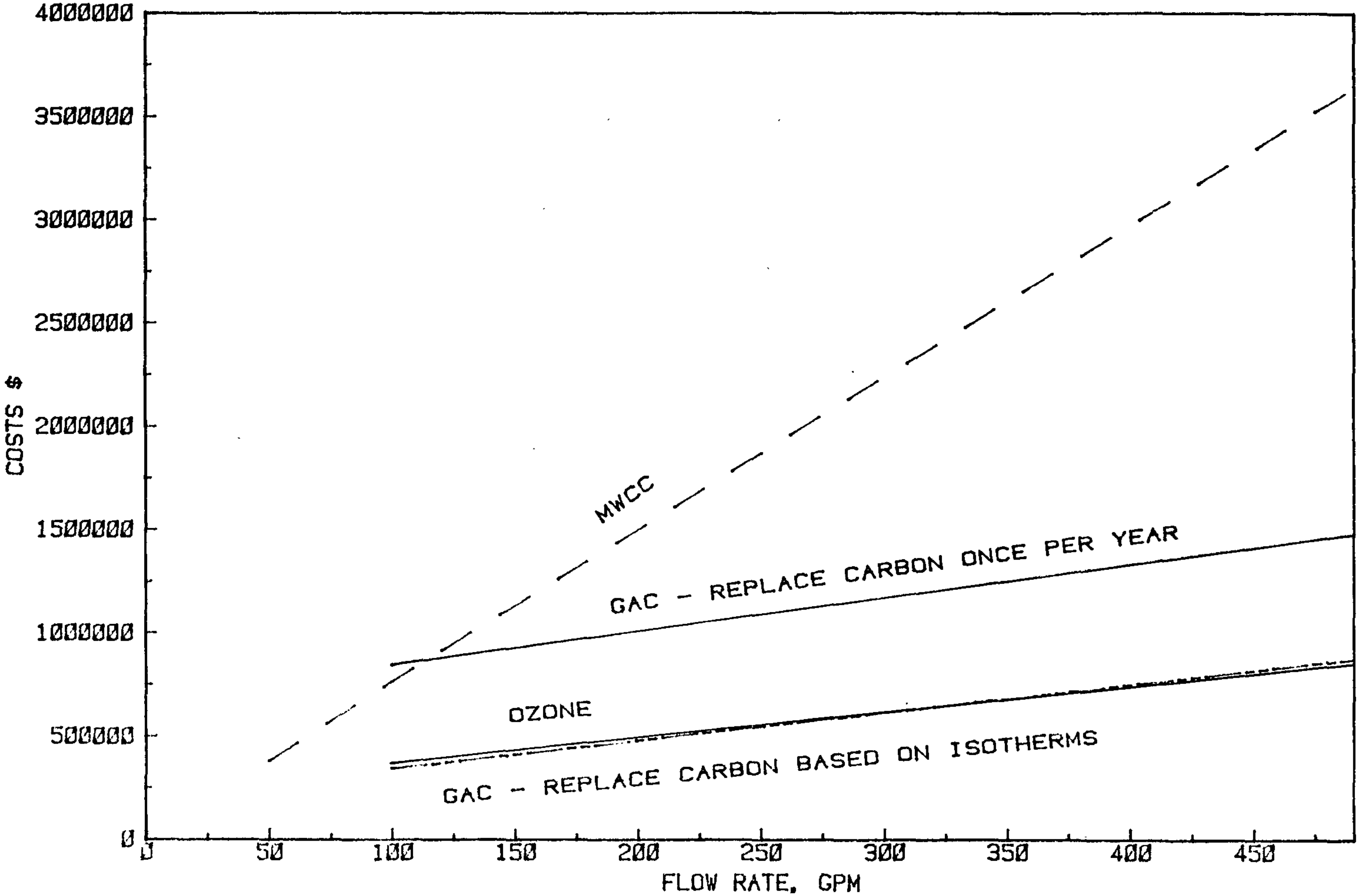

Figure F5-1 Comparison of Present Values for Treatment/Disposal

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